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Variability of Natural Background Radiation

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VARIABILITY OF NATURAL BACKGROUND RADIATION

The average annual exposure of persons in the United States to radiation from natural background sources is often said to be "about 100 millirem" whole-body dose equivalent. Though it is usually pointed out that actual exposures differ from one region of the country to another, and that the 100 mrem value is an estimate of a population-weighted average, many references include little to indicate the extent of the variations actually encountered. As a result, there is some room for the impression that this nominal 100 mrem is a sort of natural constant -- much like that of normal body temperature (37°C) -- and that any appreciable departure above this norm is associated with seriously undesirable consequences. In the present discussion it is intended, first, to describe the generally familiar range of natural background (particularly as experienced in the U.S.), and then to bring to attention some of the more fine-grained aspects of its variability. These naturally-occurring variations warrant consideration in assessing the significance of incremental perturbations of the radiation levels to which people may be exposed.

I. Natural Background

This consists of three major components: (i) Cosmic Rays, (ii) External Terrestrial, and (iii) Internal. These are described separately.

(i) Cosmic Rays

In the lower atmosphere (altitudes less than a few km) the radiation from this source is mostly provided by muons and high energy (very penetrating) electrons. There are other particles in the flux, including neutrons. The number of neutrons (at low altitudes) is small compared to the number of muons and electrons, but because of their large quality factor (Q) or relative biological effectiveness (RBE), which -- at least in UNSCEAR-1982 -- has been taken to be 10 for neutrons as compared with unity for muons or electrons, the neutrons contribute appreciably (about 10%) to the dose equivalent in tissue, even at sea level. This contribution increases with altitude, and at 3 km (9,850 ft) the neutron component contributes about 25% of the total biological dose. (More recently, the NCRP has decided that the value of Q for neutrons might lie between 5 and 20. The total level of the cosmic radiation (in rems) may, then, finally be rated somewhat differently than in some of the values used below.*)

At high altitudes (altitudes greater than about 10 km, which are accessible only to high-flying aircraft or space vehicles) there is a strong dependence of the cosmic ray flux (or dose) on the geomagnetic latitude -- the flux being many times larger at the magnetic pole than at the equator. However, on the inhabited portions of the earth's surface (altitudes less than ~5 km) the variation with geomagnetic latitude is much smaller; and for the continental U.S. (essentially all lying between 40° and 60° N geomagnetic latitude) the variation with latitude is only a percent or so. This will be ignored in the sequel.

*See Appended Note Concerning Radiation Units, (p. 26).

At any particular location on the surface of the continental U.S., the cosmic radiation may be considered as uniform in time. Though there are temporal variations associated with the 11-year sun-spot cycle, with solar flares, and with changes in atmospheric pressure and temperature, these are either of limited extent (near the surface at U.S. latitudes) or are of short duration. They may consequently be incorporated in some average value, and will not be further considered.

The significant variation in cosmic ray exposure is the variation with altitude. This results from the difference in thickness of the atmospheric blanket. On this account the tissue dose equivalent from cosmic rays at altitudes of 1, 2, or 3 km above sea level are larger than the exposure at sea level by factors of about 1.35, 2.2, and 4.0, respectively. The average cosmic ray dose rate out-of-doors at sea level is 29 mrem/yr. Since people spend a considerable fraction of their time indoors, and since structures provide at least some shielding, it has been estimated that for the U.S. the average exposure received by the population is about 10% smaller than the exposure out-of-doors. The average exposure rate at sea level has thus been taken to be 26 mrem/yr. Taking into account the distribution in altitude of the U.S. population, the average dose equivalent rate from cosmic rays has been estimated to be 28 mrem/yr. This is the number included in the assessment that the average annual exposure in the U.S. is about 100 mrem/yr.

More than 80% of the U.S. population lives at altitudes less than 0.3 km (~1,000 ft), and for these the cosmic ray dose rate is within a mrem/yr, or so, of the countrywide average. About 10 million live at altitudes $\geq$ 1 km, where the cosmic ray dose rate (out-of-doors) exceeds 40 mrem/yr. More than five million live at altitudes > 1.3 km for whom the cosmic ray dose rate exceeds 45 mrem/yr. Cities included in this group are: Salt Lake City, Albuquerque, Reno, Colorado Springs, and Denver. (For Denver, altitude 1.6 km, population 1.5 million, the cosmic ray dose rate is 50 mrem/yr). More than 100,000 live in cities -- such as Durango, Gallup, Flagstaff, and Santa Fe -- at altitudes $\geq$ 2 km, for whom the out-of-doors cosmic ray dose exceeds 60 mrem/yr. There are many small settlements in the Rockies (e.g., Silverton, Colorado, 2.8 km) at altitudes of about 3 km. In particular, for Leadville, Colorado (altitude 3.1 km) and nearby Climax (altitude 3.4 km), in or near which a total of about 10,000 persons reside, the cosmic ray dose rate would be 120-150 mrem/yr (out-of-doors).

In this same general connection, outside the U.S. there are a number of cities with large populations at quite high altitudes. These are at lower geomagnetic latitudes than apply in the U.S. As a rough allowance, in designating cosmic ray dose rates for these cities, the doses from the detailed dose-altitude curve drawn for the U.S. have been reduced by the same fraction as the sea-level doses for the relevant geomagnetic latitude. The particular dose-altitude curve used is that presented in NCRP-45 (1975). These high-altitude cities include: Johannesburg, alt. 1.8 km, population ~2 million, dose rate ~60 mrem/yr; Mexico City, alt. 2.5 km, population ~18 million, dose rate ~80 mrem/yr; Bogota, alt. 2.6 km, population ~4 million, dose rate ~85 mrem/yr; and Quito, alt. 2.85 km, population ~.75 million, dose rate ~100 mrem/yr. There is also La Paz and the Altiplano region of Bolivia.

In the Altiplano the altitude ranges from 3.5 to 4 km, and about 75% of Bolivia's total population of 6 million live in this region. In addition to La Paz at 3.6 km, population (La Paz Department -- that is, the city, plus the surrounding administrative area) 1.9 million, there is the city of Oruro at 3.7 km, Lake Titicaca and its surrounding settlements at 3.8 km, and the city of Potosi at 3.9 km, population (Potosi Department) ~0.8 million. Thus, in the Altiplano region there are 4 million, or so, people for whom the cosmic ray dose rate is in the range 150 to 200 mrem/yr.

(ii) External Terrestrial

At any location on the earth's surface persons are exposed to some flux of radiation (mostly photons) from the decay of radioactive elements contained in the soil and rocks. The main primordial sources are K-40, Th-232, and U-238; though, in the case of Th and U, the major part of the radiation encountered is provided by the radioactive daughters in their decay chains. The radiation flux at any location will vary depending on whether the soil is wet or dry, covered with snow or not, subjected to changing barometric pressure, and so forth; but these fluctuations will average out over the year. The significant variation is that applying from place to place due to differences in the local abundance of the primordial elements. Most of the radiation to which people are exposed is transmitted directly into the air from the near-surface rocks and soil as they reside in place. Almost all the radiation reaching the atmosphere originates in the topmost 25 or 30 centimeters of the soil.

On a mass basis the elements potassium, thorium, and uranium in the materials of the earth's crust are, respectively, something like two percent, and 12 and 4 parts per million. The number of atoms per gram of potassium (atomic mass ~40) is six times larger than that of thorium or uranium (atomic mass ~240). The isotopic abundance of K-40 (the only radioactive isotope of potassium) is 1.2×10^{-4}. The atomic ratios of K-40, Th-232, and U-238 in the earth's crust are, consequently, about as 4:3:1. With half-lives of 1.26×10^{9}, 1.4×10^{10}, and 4.5×10^{9} years, the number of disintegrations per unit time of K-40, Th-232, and U-238 are about in the ratio of 15:1:1. In ninety percent of the disintegrations of K-40 a β-particle (maximum energy ~1.3 MeV) is emitted, and almost all of these are absorbed in the soil close to the source. However, in the remaining 10% a γ-ray (energy 1.46 MeV) is emitted, and some of these will penetrate to the atmosphere. From the above it can be seen that in material having the average composition of the earth's crust there are about 1.5 γ-ray-emitting disintegrations of K-40 per disintegration of Th-232 or U-238 -- which are essentially equal. Th-232 and U-238 are the parent nuclei of decay series with ten or a dozen daughters having relatively short half-lives. Assuming a state of radioactive equilibrium (which doesn't always apply) each of the daughters in the series will disintegrate at the same rate as the parent nucleus. These series disintegrations release about 40 or 50 MeV of energy, but all but about 2 MeV of this energy is carried by α and β particles and deposited in the immediate vicinity of the source. About 30% of the energy carried by γ-rays is in low energy quanta (less than 1 MeV) which are strongly attenuated in the soil. In the thorium series there is a 2.6 MeV γ-ray emitted about 36% of the time, but in the uranium series there are no γ-rays with such a high energy. Thus, thorium contributes more than uranium to the terrestrial background radiation. The average concentrations of these elements in near-surface soil is somewhat lower than in the earth's crust; but in UNSCEAR-77 it is estimated that the world average radiation level at one meter above the

surface is about 40 mrad/yr: 15 from potassium, 15 from thorium, and 10 from uranium. As already suggested the actual background radiation rate from one location to another may vary considerably from this average depending on the composition of the soil or rocks nearby.

On the basis of extensive surveys the U.S. has been divided into three distinguishable regions with respect to terrestrial radiation backgrounds. These are: (i) The Atlantic and Gulf Coastal Plains Area -- a coastal belt of from one to a few hundred miles in width extending south and west from Long Island to Texas, including between 15 and 20% of the U.S. population, and within which the terrestrial radiation is said to provide an absorbed dose rate in outdoor air of between 15 and 35 mrad/yr, with a population-weighted average taken to be 23 mrad/yr; and (ii) Middle America, or The Noncoastal Plain Area, the region extending north and west from the Coastal Plains Area to the Pacific coast (except for a relatively small island around Denver and the Colorado Plateau). In this region, which includes about 80% of the U.S. population, the natural terrestrial background exposure rates range from 35 to 75 mrad, with the average taken to be 46 mrad/yr; and (iii), the Denver, Colorado Area, including some part of the East Front of the Rockies and the Colorado Plateau, in which the terrestrial exposure ranges from 75 to 140, and for which the average is taken to be 90 mrad/yr.

Much of the support for this regional breakdown is provided by the ARMS survey. ARMS refers to the Aerial Radiological Measurements Surveys of the radioactivity in the vicinity of government-sponsored nuclear facilities, conducted for the AEC between 1958 and 1963. Areas about 100 miles on a side around each of 25 locations were surveyed on a one-mile grid to map the terrestrial radiation background. About 30% of the population of the U.S. was comprised within these areas.

A range of radiation rates was observed in each area. For some of the locations, half or more of the area was noted as having rates more than ±15 mrad/yr from the mean for the area. For each area, the mean rate was taken to be applicable to the population of that area. For those portions of the country not covered by ARMS, the regional average exposures noted above were used to determine a population-weighted average of ~40 mrad/yr for the outdoor absorbed dose rate in air for the U.S.

This terrestrial radiation is mainly composed of γ-rays with an energy of one to two MeV. This radiation is attenuated by the materials in structures, and, since people spend more than two-thirds of their time indoors, and even though there may be some external dose from the building materials themselves, a factor of 0.8 has been applied to the outdoor dose in estimating the actual average exposure people receive. In addition, because of the shielding provided to the vital organs (gonads, bone marrow, etc.) by the outer tissues of the body, a further factor of 0.8 has been used in converting the terrestrial dose in air to the equivalent biological whole-body dose rate. With these factors, the population-weighted countrywide average dose equivalent from terrestrial radiation to persons in the U.S. has been taken to be 26 mrem/yr. This is the number used in the assessment that the background radiation dose in the U.S. is ~100 mrem/yr.

Surveys of background terrestrial radiation levels have also been made in other countries. Because of differences in instrumentation and procedures, not all

of these survey results are directly comparable, and not all have been carried through to the point of developing a population-weighted average. From having smaller areas the surveys of some of the countries are geographically more complete than present U.S. surveys; and, in addition, at least some have been conducted more systematically. Notwithstanding these differences, some of the values listed in UNSCEAR-82 showing the results of the surveys of about fifteen countries are indicated below. The values quoted are for absorbed dose in outdoor air in mrad/yr, which may be compared with the U.S. average of 40, already noted. The lowest average values (32-33) are for Canada, Denmark, Poland; the highest (70-90) for France, Romania, Switzerland, East Germany (GDR). In most cases ranges are given. The highest of the high range values are: Norway, 950; Italy, 435; West Germany (FRG), 315; France, 250; GDR, 235. For bottom of the range values, several were less than 10, including: Japan, Italy, FRG, France, Austria. Not to be cheated out of having something special about it, the bottom of the range for Ireland is listed as zero -- which could, of course, actually apply to a peat bog.

In a few cases, population-weighted indoor to outdoor ratios are listed. With the exception of the GDR which lists 0.8 (the same value assumed for the U.S.), these ratios are all larger than unity -- ranging from 1.65 for Austria to 1.08 for Canada. (The values for Canada are not from UNSCEAR, but from the report of an extensive Canadian survey completed in 1984.) At least on the basis of the data shown in UNSCEAR-1982, the U.S. value for indoor-outdoor ratio would appear to be one of the least well supported, being based on results from only about 270 dwellings as compared with the Norway value of 1.12 (2000 dwellings), or the FRG value of 1.36 (30,000 dwellings). Indeed, the value for this factor for the U.S. may well deserve further consideration. (In its forthcoming report, NCRP proposes to change this factor from 0.8 to 1.0.)

From this welter of data, along with data concerning the worldwide distribution of the primordial elements, UNSCEAR-82 concluded that, for external terrestrial background, a reasonable value for the global average of the absorbed dose rate in outdoor air would be about 44 mrad/yr, and that a value of 1.2 would be a suitable global average for the indoor-outdoor ratio.

The total environmental exposure to external radiation consists of the sum of the cosmic ray and the terrestrial components. For the Continental U.S., as already indicated, the population-weighted average of this sum is 28 + 26 = 54 mrem/yr. In a survey conducted in 1971 by the Lawrence Livermore Laboratory at 107 weather stations throughout the U.S. (but not including any locations at altitudes higher than that of Flagstaff, Arizona ~7000 ft), the range in this quantity was from a low of about 35 mrem/yr to a high of about 150 mrem/yr. The low values applied in southern Florida, where the cosmic component was small (sea level, less than 40° N. geomagnetic latitude), and the terrestrial component was also very low. The high values applied at Colorado Springs, Colorado (alt. ~6150 ft) which has fairly high components, both cosmic and terrestrial; and Bishop, California (alt. ~4150 ft) with a moderate cosmic component, but very high terrestrial. Flagstaff, Arizona, with the highest cosmic component of the locations included in this survey, had a rather low terrestrial component, and a total exposure to external radiation of only about 90 mrem/yr. In Hawaii (near sea level, and only 20° N geomagnetic latitude) the cosmic component was smaller than in Florida and the terrestrial components were also very low; so that external radiation provided about 30 mrem/yr for the locations monitored. In the reports examined, no measurements were given

of the terrestrial component of external radiation for the high-lying settlements in Colorado (>7000 ft altitude). There is, however, a general tendency for the external terrestrial radiation at such locations to be high -- in part, no doubt, because of the presence of rock near the surface, or of the exposure of bare rock. It therefore seems likely that among these settlements, which already have a cosmic ray exposure in excess of 100 mrem/yr, there will be some for which the total environmental exposure is >200 mrem/yr.

(iii) Internal

The exposures from internal sources of radiation may conveniently be considered in three classes: (a) that from normal constituents of the body (principally potassium); (b) that from radionuclides lodged in the body (uranium, etc.); and (c) exposures from inhaled radionuclides (radon and its daughters).

(a) The concentrations of the normal constituents of the body (such as H, C, or K) are maintained at fairly constant levels by the body's state of physiological equilibrium. They are consequently largely independent of such factors as diet or geographical location. In the absence of temporary man-made perturbations -- such as tritium releases, nuclear explosions, and so forth -- the isotopic composition of such elements in the body will be the same as that in the biosphere.

Cosmic rays provide a steady source of a large variety of radionuclides -- mostly produced at high altitudes. These mix with the lower atmosphere and other components of the biosphere and the deep ocean reservoir, and have established and maintained for a very long time a stable concentration in the various parts of the environment. The concentration of any particular cosmogenic radionuclide in any particular component of the environment depends strongly on the half-life of the nuclide (along with other factors, such as solubility).

In the biosphere (the lower atmosphere, surface waters, plant life, etc.), the four most abundant cosmogenic radionuclides are C-14, Na-22, Be-7, and H-3. Except for Be, these elements are essential constituents of the body. The total internal dose delivered by these radionuclides is about 1 mrem/yr, and is almost all provided by C-14; being, in particular: C-14, ~1; Na-22, ~0.02; and H-3, 0.001 mrem/yr. (Though not a body constituent, Be-7 may be ingested or inhaled, and is estimated to provide an internal dose of about 0.008 mrem/yr.) The total dose from all other cosmogenic nuclides is thought to be less than .001 mrem/yr.

Potassium is an essential constituent of the body, with an abundance of about 2 grams per kilogram of total body weight. Strictly speaking, the 2 gm level applies only to young males (age ~20) and falls essentially linearly with time over the next 60 years to about 1.6 gm. In females, after age 20, the potassium concentration at all ages is only about 75 to 80% of that in males -- in part, possibly, because of the difference in proportion of adipose tissue in which the potassium concentration is relatively low (only about 0.5 gm/kg). There is an appreciable variation in potassium concentration from one organ of the body to another (~4 gm/kg in red marrow, 2 in testes, 0.5 in bone) and a corresponding variation in doses to the different organs. However, for an assumed average concentration of 2 gm/kg body weight, the whole-body dose equivalent has been estimated to be ~18 mrem/yr.

Essentially all the β-particles from the decay of K-40 will be absorbed in the body; but more than half of the γ's will escape. Because of this, each person carries a small radiation field around with him. This, no doubt, is the basis for the jocular comment that there is some hazard (from radiation) in sharing a double bed. The hazard, of course, is not very great, being on the order of only a tenth of a mrem/yr in bed. However, since a nearby body would screen about 10% of the solid angle from the normal external terrestrial radiation of ~30 mrem/yr, it might better be said that sharing a double bed has a favorable effect.

After K-40 the most prominent nonseries primordial radionuclide is Rb-87. This nuclide emits only β-particles (maximum energy 0.27 MeV) so it is significant (if at all) only as a source of internal dose. Considering the factors of elemental abundance in the earths crust, isotopic fractions, half-lives, and energy per disintegration, the dose from Rb-87 would be about fifteen times smaller than that from K-40 -- provided the concentration in the body relative to that in the earth's crust should be the same. From measurements of rubidium in the body it has been concluded (UNSCEAR-77) that the dose rate from Rb-87 is about 0.4 mrem/yr. This is about forty times smaller than that from K-40.

In addition to K-40 and Rb-87, there are about twenty other nonseries primordial radionuclides in the material of the earth's crust. Considering their elemental abundances, isotopic fractions, etc., their rates of energy release per gram of terrestrial material range from a few percent down to many orders of magnitude smaller than that from Rb-87. The contribution of these to internal dose may consequently be ignored.

In summary, the dose rate from radioactive constituents of the body (K-40, C-14, H-3, etc.), is from 18 to 20 mrem/yr.

Finally, in this discussion of natural backgrounds, it is not intended to discuss the effects of the testing of nuclear explosives except as these may have affected items in the natural background. The immediate effect of nuclear testing (from the mid-1950s to 1963) was to release in the atmosphere large quantities of radioactive fission fragments (such as Sr-90 and Cs-137) which were not otherwise present in the environment. These will decay (or have decayed) to inconspicuous levels providing the present ban on testing in the atmosphere continues. As to the isotopes already considered in connection with natural background, the effects were as follows: for Rb-87 -- even though this is a direct fission product -- the amount added was much less than one percent of the natural abundance of this nucleus in the upper millimeter of the earth's crust. Eight of the other primordial radioisotopes are also direct fission fragments. For these, also, the contribution from testing was a very small fraction of the abundance of these isotopes in the topmost layer of the earth's crust. K-40 is not a fission product, so there was no effect on that.

C-14 is not a fission fragment, but it is formed by the capture of neutrons in the nitrogen of the atmosphere. H-3 (tritium) is also not a fission fragment, but is a residue of the burning of thermonuclear fuel. The inventory of C-14 in the biosphere was approximately doubled as a consequence of weapons' testing. The previously ascribed one mrem/yr from this source could have been raised to something between 1.5 and 2 mrem/yr. This incremental effect will decrease much more rapidly than it would merely as a result of the radioactive decay of C-14

(half-life about 5,700 years) because of the process of equilibration with the contents of the deep ocean reservoir. This process is believed to proceed with a mean life of about 7 years, or so. The present (1986) level of C-14 in the biosphere is about 20% larger than the "natural" level of C-14.

It has been estimated that the global inventory of tritium (H-3) was increased by a factor of between several hundred and a thousand by the nuclear explosions conducted in the atmosphere prior to 1963. With a half-life of 12.3 years, the amount of injected tritium will by now have been reduced by a factor like 5; but, it still completely masks the effect of "natural" tritium, and will continue to dominate for the next hundred years or so. Even at that, of course, it is a rather small term in the total exposure to natural radiation.

(b) Apart from the radioactivity associated with essential constituents of the body, there is some internal dose resulting from the ingestion of "foreign" radionuclides in the environment. The amount of these is not homeostatically controlled, but depends on their concentration in materials (air, water, and food) taken into the body. The items of particular concern here are the parental thorium and uranium and some of their daughters, such as radium. Their gaseous daughter, radon, will be discussed separately later.

Though the amount of these elements taken up in the body was once, no doubt, rather directly related to the concentration of these elements in the local environment, that is no longer so much the case. It is still true that some of the underground water in Iowa and Illinois, as well as at other locations in the country, has an unusually high radium content; but an increasing fraction of such water is now treated before it reaches a consumer. More significantly, with the greatly increased use of canned and packaged foods (which may be processed anywhere in the country) and the countrywide distribution system for produce of all sorts, the U.S. food supply has become homogenized to a very large extent. Consequently, in discussing the uptake by ingestion of the series radionuclides it seems appropriate to use the average values estimated for the U.S. Quite apart from the (relatively) straightforward matter of assessing the average uptake of uranium and thorium (and daughters), the matter of correlating this with a whole-body equivalent dose requires composing a number of radically different effects: the ingested radionuclides spend some time in the stomach, some time in the bloodstream, and some end up deposited in the gonads and on the bone surfaces. The amount of thorium ingested is probably about the same as that of uranium; but the retention of thorium in the body is very much smaller. As a consequence, most (80 or 90%) of the internal dose from the series radionuclides is provided by uranium and its daughters.

In the following discussion the estimates compiled in the 1975 report, NCRP-45, will be presented; but at the end of this section on internal exposure some comparison will be made between these estimates and the newer (1986-87) estimates being considered by the NCRP. From NCRP-45, then, the ingestion of the primordial series radionuclides results in a whole-body equivalent dose rate of about 7 mrem/yr. Uncertainties and differences which could readily affect this estimate would not greatly affect the estimate of the total dose from internal sources since this is dominated by the dose from K-40, which is about twice as large as that from uranium. Thus, with the exception of the dose resulting from inhaled radon (and daughters), the dose equivalent rate from internal sources is about 26 mrem/yr -- ~20 from K-40, and 7, or so, from

uranium, etc. This is the number assumed in the assessment that the average dose to persons in the U.S. is about 100 mrem/yr.

(c) The main additional source of internal radiation is that resulting from the inhalation of radon and its short-lived daughters. Radon appears at nearly the same rate in both the uranium and thorium decay series, and is the only gaseous element in these series. In the uranium series, the isotope Rn-222 is an al, a-emitter with a half-life of 3.8 days. This allows time for an appreciable fraction of the radon formed near the surface to migrate into the atmosphere and to be carried about by the wind. In contrast, the isotope Rn-220, which appears in the thorium series, has a half-life of only 55 sec, so that it does not succeed in migrating from the soil to an extent which warrants consideration in comparison with the 3.8-day Rn-222.

Radon is an inert monatomic gas -- one of the "noble" gases, which engage in few, if any, chemical reactions. Once released to the atmosphere these atoms move freely about and the products of their decay appear as single atoms and attach themselves either to some molecule in the air or to an aerosol particle and thus remain suspended in the air for a considerable time. Radon decays by α-emission; and if this occurs while the radon atom is still suspended in air there is no direct effect on human exposure. The immediate daughters of Rn-222 (Po-218, Pb-214, Bi-214, Po-214) have short half-lives (from 0.16 msec, to 27 min) and their decays are also likely to occur while the atoms are still suspended in the air. The first and last of these decays are by α-emission; so that, again, there will be no direct effects on exposure to humans -- unless, of course, the original radon atom, or one of these daughters had been taken into the body by inhalation and the energy of the subsequent decay were deposited there. However, the second and third daughters are β-emitters, and their disintegrations are accompanied by a large fraction of the gamma-ray energy appearing in the uranium decay series. Thus, even if these disintegrations occur while the daughters are still suspended in the air they would provide some external exposure to humans -- though not a very important component from radon concentrations normally encountered in outdoor air. The (temporary) end-product of this group of decays is the (relatively) long-lived Pb-210 (21 years). This undergoes two β-decays followed by the emission of an α-particle, which terminates the uranium series in the stable isotope Pb-206. There are essentially no gammas associated with the decay of Pb-210; so this isotope contributes only to internal exposure. That could result either from the inhalation of air in which Pb-210 were still present after the decay of Rn-222, with some fraction of the Pb-210 being lodged in the body, or from the ingestion of plant-life growing on soil in which the Pb-210 had been deposited. The former is by far the more important route for exposure to radiation from Pb-210.

Very little radon is emanated from the surface of the ocean, and on this account the concentration in coastal air is low and variable -- depending on whether the air is moving from inland or from the sea. In the continental air mass, the level of radioactivity is about 150 pCi (pico-curies: 10^{-12} Ci) per cubic meter. A large fraction (>2/3) of the radon inhaled is exhaled before it decays, but the solid radon daughters (the 21-year Pb-210 and the 140-day Po-210) attach to the surfaces of the pulmonary tract -- and particularly to the walls of the hair-like passages in the segmental bronchioles. The dose rate to the tissues of the lung from this cause has (in NCRP-45) been estimated as being about 90 mrem/yr, and to the bronchial epithelium about 450 mrem/yr.

Using the weighting factor recommended by the ICRP (whole-body dose equivalent at 0.12 times the dose to the lung tissue), the whole-body equivalent dose from exposure of the lung tissue would be about 11 mrem/yr. If one applies the ICRP-recommended weighting factor of 0.08 to the dose to the bronchial epithelium, this would add an additional 36 mrem/yr to the whole-body dose equivalent. Adding to the 80 mrem/yr already identified (28 cosmic, 26 external and 26 internal), we have an average natural background exposure for persons in the U.S. of rather more than 100 mrem/yr, without taking into account the proposed revision of the indoor/outdoor factor from 0.8 to 1.0, which would raise the external component from 26 to 32 mrem/yr.

Up to this point the exposure to inhaled radionuclides (radon, etc.), has been described only in terms of persons breathing outdoor air. In fact, of course, people spend a major fraction of their time indoors, and the radon levels in dwellings may be quite different (usually higher) than the radon levels out-of-doors. Radon seeps into dwellings from the soil in which the basement is embedded, from the materials of construction -- such as cinder blocks -- and, because the rate of exchange of air in dwellings is intentionally much smaller than the rate of exchange of air outdoors (in houses weatherproofed for energy conservation, a great deal smaller), the radon concentration in indoor air may run much higher than in the ambient air outside. The effects of this have not been considered here as part of the "natural background," since they are, in fact, technologically enhanced and could (in principle, at least) be controlled. They do, nevertheless, provide an additional source of radiation to which the population is exposed. Some (quite partial) surveys have been conducted. These do not yet begin to be adequate to establish an average level for indoor radon exposure for the U.S. From the surveys which have been made examples have been found in which the indoor radon levels were ten, or more, times larger than the continental outdoor average. Such a level would imply an equivalent whole-body dose larger than the average already identified by a hundred -- or even more -- mrem/yr.

As stated earlier, the components of the dose equivalent rates from natural background radiation as given above are derived from the data provided in NCRP-45. In its 1982 report the UNSCEAR directed much more attention to radon than it had in previous reports; saying, in particular: "Inhalation is now recognized to be the most important pathway," -- and "on average about one-half the effective dose equivalent from natural sources of radiation is now calculated to be due to the presence of radon in the air inside buildings."

In the January 1987 draft of a forthcoming NRCP report, the dose equivalent values for cosmic radiation, terrestrial gamma radiation, and the internal dose from cosmogenic radionuclides and K-40 are changed very little. But there are marked changes in the components where the exposure is provided primarily by α-radiation: the uranium contribution to internal radiation, and, most particularly, the dose attributed to inhaled radon. These changes were in part occasioned by the increase from Q=10 to Q=20 for α-radiation; but they were also affected by new data showing higher concentrations of Pb and Po-210 in bone, by higher estimates for the tissue dose from radon decaying in the body, and particularly by including some allowance for the higher level of radon indoors as compared to outdoors. More specifically, the contribution of uranium to the internal exposure is now being rated as about 10 to 15 mrem/yr whole-body dose equivalent (rather than the value of about 7 noted above); and the dose rate proposed for the bronchial epithelium is 2,450 mrem/yr (rather

than the 450 suggested in NCRP-45). Applying the weighting factor of 0.08 to the dose to the bronchial epithelium would add about 200 mrem/yr to the whole-body dose equivalent. In summary, the draft version of the forthcoming report provides an estimate of the total average annual exposure to a member of the population of the U.S. from sources of natural background radiation of 300 mrem.

It should be noted that there is some continuing controversy about the proper weighting factor for the whole-body equivalent of dose to the bronchial epithelium. In addition, as already mentioned, the data to establish a country-wide average of indoor radon concentration is still far from complete -- though additional surveys on this point are in progress. For both of these reasons the estimate of the contribution from the bronchial epithelium to the total dose must be regarded as still in question.

II. Celebrated Hot Spots

There are locations in which the natural background of terrestrial radiation is much higher than those so far referred to. A particularly notable one is the Kerala Coast. (The state of Kerala is on the west coast of India near the southern tip.) In a narrow strip, extending 100 miles, or so, along the beach, numerous patches of monazite sand are exposed. (The mineral monazite consists of highly insoluble phosphates of cerium and other rare earth elements in various proportions, usually accompanied by some thorium and, on occasion, small amounts of uranium, and their daughters.) The most concentrated deposits are found in a 30-mile section of the strip; and there the monazite contains from 8 to 10.5 percent thorium by weight -- the highest known in the world. About 70,000 persons live in this section. There is, of course, considerable variation in the external terrestrial exposure received by the people residing in this region (some of the dwellings -- which are mostly made of coconut straw and wood -- being located directly on patches of monazite, and some not; some residents being employed outside the high background area, while others spend most of their time near home). However, on the basis of radiometric surveys, the average exposure to terrestrial radiation for the 70,000 persons in the region has been estimated to be about 380 mrem/yr. For about 17,000 persons the exposure has been estimated to exceed 500 mrem/yr. It exceeded 1,000 mrem/yr for more than 4,000 persons; and it exceeded 2,000 mrem/yr for about 500. People have been living in this part of India for hundreds of years. It is very densely populated, and it would seem unlikely that there has been any large influx of people from outside for a long time. In all probability most of the present residents have generations of ancestors who also lived in this region. Some preliminary epidemiological studies have been made, and more are planned. Still -- at least as reported up through about 1980 -- no statistically significant evidence has been found of effects resulting from the unusually high background radiation to which the population of the Kerala Coast has been exposed.

Impressive deposits of monazite sands also occur on some of the beaches of Brazil, about 200 miles northeast of Rio de Janeiro. In particular, in the town of Guarapari -- which has a resident population of 12,000 persons, and a summer tourist population of 30 to 40 thousand -- it has been estimated that the average annual exposure rate to external terrestrial radiation in the town is about 550 mrem/yr. Along the beach of this health resort there are patches

of "black sand" (particularly favored by the tourists) on which the radiation levels are from five to ten times higher than in the streets of the town.

There is a small agricultural area in China about 100 miles southwest of Canton in which an appreciable concentration of monazite has been deposited by alluvial action. About 80,000 persons reside in the high-radiation area, and over 90% of these have had six or more generations of forebears who lived in the same area. There are similar long-established villages at distances of only 10, or so, km where the concentrations of U and Th in the soil are from 3 to 10 times smaller; and these have provided a control group. Though the external terrestrial radiation level is four times greater in the high-radiation area than in the control area the total whole-body exposure (including cosmic and internal components) is only 2.4 times greater, being about 230 and 95 mrem/yr, respectively. Extensive medical surveys have been made of the two population groups to obtain data concerning such factors as morbidity and mortality rates from malignancies, spontaneous abortion rates, and the incidence of hereditary and congenital diseases. In addition, more than 20,000 individuals from each group were examined to check for differences in chromosomal aberrations, leukemia, and measures of growth and development. In a number of instances the results for the two groups were essentially identical, and in no case was a statistically significant difference observed. Although no appreciable effect was found the Chinese Radiation Research Group which conducted the studies concluded that the size of the population group was too small to show minor increments of detrimental effects at such low doses.

In addition to the monazite beaches there is a region in Brazil with very high terrestrial background radiation in a distinctly different geological setting. This is a volcanic area about 200 miles west (inland) from Rio and extending north from the city of Pocos de Caldas to Araxá where there are intrusions containing minerals having close to two percent thorium oxide and over one percent uranium oxide. Radiation levels up to twice those noted in the streets of Guarapari have been measured near Araxá, and on a small uninhabited hill -- the Morro do Ferro -- near Pocos de Caldas absorbed dose rates in air up to 24 rads/yr have been reported. No large population groups appear to be exposed continually to the very high radiation background in this region.

In France locations providing absorbed dose rates in air of about 1.75 rads/yr are not uncommon, and the discovery of a quite small area providing a rate of over 80 rad/yr has been reported. There are also locations in Paris where one may receive a biological dose of up to 350 mrem/yr. Though no one actually lives in St. Peter's Square in Rome, many people spend appreciable time there, where it is reported that the paving stones provide up to something like 400 mrem/yr. The Fichtelgebirge is a granitic mountain near the northeast border of Bavaria. There are several towns or villages on the slopes of this mountain. On the streets of these villages the terrestrial γ-ray exposure ranges up to more than 500 mrem/yr -- the highest known in the FRG.

In Grand Central Station in New York City -- which was built with granite from the Millstone Quarry in Connecticut -- there are locations where the external terrestrial dose rate is about 525 mrem/yr. Stone from the same source was used in constructing the foundation for the Statue of Liberty in New York harbor, and this also provides a high radiation exposure. (While it was operating -- from about 1740 to 1960 -- the Millstone Quarry was a favored source of building material since it was immediately adjacent to the shore, and rock could be

transported readily to locations on the East Coast. The radiation exposure of persons working in this quarry must have been quite high.) High radiation levels (absorbed dose rates in air up to 150 mrad/yr, or so) can also be found in other granitic regions of New England, and, indeed, wherever else similar rock may be found at the surface. A different setting for high terrestrial background radiation is presented by the phosphate deposits in Florida. From this appreciably uraniferous material terrestrial background radiation levels of absorbed dose rates in air up to 150 mrad/yr have been observed. Deposits of phosphate rock occur throughout the world. Among the major phosphate-producing areas the deposits in South Carolina, Wyoming, and some of those in Brazil have higher concentrations of uranium than those in Florida, while a number of others are comparable to the ones in Florida.

The remaining type of situation resulting in unusually high exposures to natural background radiation (excluding the circumstances affecting underground miners) has to do with water. In the ionization states most usually occurring in natural settings, radium is much more soluble and mobile than either uranium or thorium. On this account water -- and particularly warm water -- flowing through beds of sandstone or fractured granitic rock may accumulate concentrations of radium very much higher than the concentration in the material through which the water has been flowing. At locations where such water may emerge to the surface one has the makings of a "radium spring," or -- where the neighboring population is sufficient to support it -- a "spa."

Locally notable "hot springs" occur in all parts of the world. Many of these became famous as "health resorts" long before the existence of radium was known, and before measurements of levels of radioactivity were ever considered. Of interest here is the fact that not only do some of the "waters" carry a level of radioactivity which would now be regarded as distinctly unhealthy, but the radon decay product of the radium in the water is released to the atmosphere and provides an unusually high level of exposure to the population in the neighborhood.

There are reports concerning a few notable radioactive hot springs. For example, the springs at Tuwa, a village in India about 200 miles north of Bombay, have a high concentration of Ra-226. In the air close to the main spring at Tuwa, the γ-ray dose (from the short-lived radon daughters) has been reported to be about 10, or more, rad/yr. At a distance of about a dozen kilometers (and several villages) downwind, this exposure rate falls to ~750 mrad/yr. Similarly, in the city of Ramsar, a resort on the Caspian coast of Iran, population >10,000, there is an area of a few square kilometers around the radium-bearing springs (which emerge in downtown Ramsar) within which levels of absorbed dose in air have been measured ranging from 1.75 to over 40 rads/yr.

The springs at Badgastein, Austria (about 50 miles south of Salzburg) have received the most extensive and detailed studies of radioactivity, both as to the "waters," and as to the surrounding neighborhood. This famous spa has been known as a "watering place" for more than six hundred years. Already in the 18th century several thousand persons travelled there each year for treatment. Over the centuries many accounts have been written (including one by Paracelsus, printed in 1562) describing the therapeutic effects of the baths at Badgastein. Badgastein gained in popularity, so that by 1940, 30,000 visitors were reported, and by 1970, about a million baths per year were administered. By this time, also, about 300 hotels were said to be operating in the region to accommodate

visitors, and the permanent population of Badgastein and environs was about 12,000.

In 1904 the presence at Badgastein of "emanation" (as radon was then known) was established by P. Curie and colleagues. Subsequent studies have determined that, although the amounts of U, Th, and Ra in the spring water are not exceptionally high, the Rn-222 content is outstanding. For most of the visitors, or spa patients taking only a few treatments, the dose received is low (from a few, to a few tens, of mrem). For patients taking a "whole cure" (a dozen 2-hour sessions in the "thermal gallery" in which the Rn-222 concentration is 3,000.pCi/l), the dose to the lung tissue is about 900 mrem -- and several times more to the bronchioles. By inhalation of Rn-222 the 5 or 6,000 permanent innabitants of Badgastein proper -- where the springs are located -- receive from 0.7 to 1.5 rems/yr (in lung tissue). The bath attendants, other personnel connected with the treatment facilities, and, particularly, the doctors attending patients in the "thermal gallery" (a group of only a few hundred persons) receive from about two, up to several tens, of rem/yr (to lung tissue) -- or did receive such exposure until about 1970 when some corrective measures are said to have been placed in effect. (The dose levels reported in this and previous paragraphs are all in the "old scale" using Q=10 for alpha particles.) Surveys have been made to compare the general health of residents of Badgastein with that of groups living in similar circumstances -- but not having any enhanced radiation exposure. These resulted in the conclusion that the longevity of the Badgastein residents was not less, and the incidence of cancer was not greater, than that for the other population groups. As of 1972 studies to identify possible radiation-induced anomalies in cells had led to the tentative conclusion that at dose levels up to somewhere between 0.3 and 1.0 rem/yr, there was no clear evidence of cell damage. For doses larger than somewhere between 0.3 and 1.0 rem/yr, there was an increasing incidence of (for example) broken chromosomes. Presumably, such studies at Badgastein have by now been extended.

There are many other well-known hot springs, or mineral springs, which have not been discussed at recent symposia on high natural environmental radiation. This could be because they have been studied, and found not to have radiological features of interest; or because specific studies have not yet been made.

Among these are the springs at Bath, in southwest England -- a spa well-known and used since Roman days. At about the same time as Curie made his findings at Badgastein, J.J. Thomson (who discovered the electron in 1897) reported the existence of copious amounts of "emanation" at Bath, and suggested that the salubrious properties of the waters there might be due to their radioactivity. With respect to the waters at Saratoga Springs, New York -- though it has been pointed out that the waters bottled and distributed from there come from a spring having low to moderate radioactive content -- some of the long-time residents, preferring the water from a different spring having several hundred times the radium content recommended (since 1962) by the NCRP as "maximum permissible," have been making regular use of this more radioactive water for periods up to 50 or 60 years without any apparent deleterious effects. Reports concerning the radioactive properties (if any) of the springs in Vichy, France (famous since Roman times) or at Hot Springs, Arkansas, or Warm Springs, Georgia, and many other locations could also be interesting.

III. Some Local Surveys

Partial results from four sets of observations of environmental radiation are described. The measurements reported are the sum of contributions from terrestrial gammas and cosmic radiation to the exposure in air -- mostly in outdoor air. The data considered were drawn from: (i) reports of EML -- the Environmental Measurements Laboratory of the U.S. DOE; (ii) NUREG-0837, the quarterly reports of the NRC TLD Direct Radiation Monitoring Network; (iii) the annual reports of the Los Alamos Environmental Surveillance Group; and (iv) readings taken in the course of a mini-survey made by the author in downtown Washington, DC, in the early summer of 1985.

(i) EML Data

Over many years members of the staff of the EML (initially the AEC's Health and Safety Laboratory -- HASL) have studied a very wide range of aspects of environmental radiation. Here, only three particular projects are referred to. The first of these is a program initiated in the fall of 1971 to monitor continuously the exposure level in outdoor air. Thermoluminescent dosimeters (TLD's) were set up near four residential locations in the suburbs of New York City, and were monitored on a monthly basis. The sites were (roughly) in directions west, north, and east, and at distances between about 15 and 30 miles, from Central Park. These locations are all close to sea level (cosmic radiation exposure about 29 mrad/yr in outdoor air) and in the Coastal Plain region (average exposure to terrestrial radiation previously said to be 23 mrad/yr). On this basis the exposure at these locations would be about 52 mrad/yr.

The 10-year average exposures measured ranged from 53 to 60 mrad/yr -- in acceptable conformance with the nominal regional value. The annual averages at a given site were observed to fall in the range (maximum-minimum/minimum) of only about 10%, but the measurements for a given month showed differences of as much as 40% from one year to another at a given site. Such differences were attributed mainly to differences in the annual snow cover and rainfall.

One consequence of such variability is that it may be difficult to obtain a precise measurement of the size of some increment in exposure level (such as might result from reactor operation or other non-natural source of radioactivity) -- at least on the basis of TLD readings, and particularly if the increment is small compared to the background. The TLD registers the sum of the incremental and background exposures integrated over some period of time. To assess the increment it is necessary to subtract the background contribution from the total reading. Since the background may vary, and cannot be read separately, the background contribution will have to be assumed on some basis, and this may leave room for considerable uncertainty in the actual size of the increment. This could, of course, be greatly improved by the use of more elaborate detectors, such as a spectrometer which could identify source isotopes; but such equipment is not attractive for use in field monitoring.

The second EML project to be mentioned here is their continuous monitoring over several years of the natural radiation exposure rates at Shoreham, NY, and the EML station at Chester, NJ. The Shoreham site is on the North Shore of Long Island, at sea level, and in the Coastal Plain region for terrestrial radiation. Chester, NJ is a little more than 90 miles west of Shoreham, at an altitude of

about 750 ft, and near the eastern fringe of the Middle America terrestrial region. The annual average exposure rates measured by EML were 59 mrad/yr at Shoreham and about 109 mrad/yr at Chester. The cosmic ray components will have been 29 and about 31 mrad/yr, respectively, so that the terrestrial components were about 30 and 78 mrad/yr. The terrestrial level at Shoreham is well within the range (15 to 35) previously ascribed to the Coastal Plain region, but the level at Chester is just above the range (35 to 75) ascribed to the Middle America region. The 50 mrad/yr difference in exposure rates is smaller than many of the variations identified earlier, but it is of interest to find that it applied between locations which would not normally be thought of as widely separated nor in different geographical provinces of the country. Actually the 50 mrad transition is much sharper than indicated by the Chester-Shoreham comparison since two of the residential sites discussed above, for which the average annual exposures measured by EML were within one mrad of that reported for Shoreham, are less than 25 miles east of Chester.

The third EML project considered is their sponsorship of a series of International Intercomparisons of Environmental Dosimeters. Eight such exercises were held between 1974 and 1986 with participants from 130, or so, laboratories from over 30 countries. The TLD exposure readings were compared with each other and with control readings on continuously monitoring high pressure ionization chambers. Many factors contributed to differences in the results obtained in the intercomparison. These included effects from differences in packaging -- where both wall thickness and ambient temperature of the luminescent element affected the readings; differences in calibration methods; in spectral response -- as for example between terrestrial gammas and cosmic radiation; problems with signal loss, or "fading," for some phosphor types; and a few others.

The conclusion from the intercomparison series was that over 85% of the participants obtained results within ±30% of the delivered exposures and that about half the results were within ±10%. Some of the test exposures included in this observation were at higher levels than typical environmental levels, and in general the percentage spreads in the readings are somewhat larger at lower exposure levels. This is partly because the corrections which must be applied for exposure during transportation and storage of the TLD's constitute a larger fraction of the total. For this reason, also, TLD readings of background over short periods -- much less than a month, say -- will not be very accurate. It follows that exposures reported from different countries or different laboratories may not be fully comparable. However, it may be expected that surveys made by a single organization using standard procedures and equipment will provide fairly good data on the differences in exposure levels from place to place or from time to time.

(ii) NRC Survey of Nuclear Power Plant Sites

Since August 1979 (a few months after the accident at TMI-2); the NRC has maintained a network of TLD's around every licensed nuclear power plant site in the country, both those under construction and those in operation. In each case about 40 detectors are emplaced in a reasonably uniform azimuthal distribution at various distances from the plant -- nominally, 16 within 2 miles of the plant, but outside the plant boundary, 16 between 2 and 5 miles, and 8 between 5 and 20 miles from the plant. The detectors are collected every three months and

replaced with fresh ones, and the readings from the exposed detectors are reported in the quarterly series NUREG-0837.

The cosmic component is uniform over the extent of the array at any particular site, so that any variation in a single array will be entirely due to differences in the terrestrial background -- unless effluents from an operating plant should lead to higher readings on detectors close to the plant in the downwind direction. However, there is little evidence of a general pattern of this sort in the data collected, and for plants still under construction there is no such consideration. Indeed, there are twice as many instances in which the highest zonal average is for one of the two outer zones rather than for the innermost zone. At two of the sites detectors unusually close to the plant also had the highest readings in the array. The data from these stations has been ignored in the following, as has the data from a few other stations which provided readings which were obviously erroneous -- such as indicating levels smaller than that of the cosmic component alone, or levels which, for one particular quarter, were much higher than that for any other station in the array while, for other quarters, the level at the same station was not outstanding. In NUREG-0837 the exposure rates for the absorbed dose in outdoor air are listed in terms of mR/quarter; but these are converted below to mrad/yr.

In the 4th quarter of 1983 arrays were operated at 69 sites, but from trouble in collecting the data needed to normalize the detector readings at 12 of these sites corrected data are available for only 57 sites. The average exposure rate for the 57 sites during this quarter was about 66 mrad/yr -- in reasonable agreement with the 70 mrad/yr (30 cosmic plus 40 terrestrial) previously identified as the country-wide population-weighted average value for the exposure rate in outdoor air. The site average rates ranged from a high of 108 mrad/yr (Fort St. Vrain) to a low of 42 mrad/yr (Catawba). The highest (non-anomalous) reading for a single station was 135 mrad/yr at a location 13 mi from Fort St. Vrain.

Of particular interest in the present discussion is the range of readings among the various detector stations within the limited extent of a single array. In the 4th quarter of 1983 the average over the 57 sites of the difference between the highest and lowest exposure rates recorded at each site was 34 mrad/yr. Amongst the sites this difference ranged from a low value of 16 mrad/yr to a high of 59 mrad/yr. This maximum spread was between two stations in the Surry array where one station, 3.7 mi from the plant, recorded a rate of 39 mrad/yr, while the other, 11 mi from the plant and 13 mi from the first, recorded a rate of 98 mrad/yr. Of course, the rates recorded at these stations -- as for almost all stations -- change from one quarter to the next as do the differences between them; but during the whole of 1983 the difference in exposure at these two stations was 42 mrad. This same difference in exposure for the year also occurred between two stations, only 1.5 mi apart, in the array at North Anna. While the maximum spread within an array was found at Surry in the 4th quarter of 1983, for the other quarters of the year (first through third) the maximum spreads were as follows: McGuire (56), Surry (57), and North Anna (54). Along with these maximum spreads, in the NUREG-0837 data for 1983 differences in exposure rates of more than 40 mrad/yr between stations in a single array were recorded at more than two dozen sites. In half of these instances the stations involved were less than 10 mi apart. Except for the Far South East (as in Florida, for example, where the terrestrial background on undisturbed land is generally too small to allow room for variations as large as 40 mrad/yr) these arrays had an

essentially country-wide distribution: from the Pacific coast, through the mid-continental region, to the Atlantic.

Finally, considering that in the NUREG-0837 survey there were nominally 8 stations between the 5 and 20 mile circles around the plant, on the average each station reported on an area of about 145 square miles. There is no reason to suppose that the extremes in the naturally-occurring exposure rates within the arrays would necessarily be picked up in this survey.

(iii) The Los Alamos Survey

For many years the Environmental Surveillance Group of the Los Alamos Laboratory has monitored a large number of locations in the technical areas of the Laboratory, and also in the surrounding neighborhood, for the presence of a long list of possible radioactive and chemical contaminants in the air, soil, and water. As a part of this operation they have maintained an array of TLDs to monitor the cosmic ray and terrestrial radiation background. A number of these TLD stations are outside the perimeter of the technical area at locations where normal Laboratory operations would not affect the readings of the dosimeters. Seven of these outside stations are deployed in the townsite; and these are all in generally similar (mesa-top) terrain, and are all at an altitude close to 7250 ft (2,200 m). They are all located within an area somewhat less than 7 square miles, and the extreme distance between any two of these stations is only 3.5 miles. These seven stations thus constitute a rather compact array. The measurements reported are believed to be within 4 percent of actual levels.

The TLDs register the sum of the absorbed dose in outdoor air from the cosmic and terrestrial backgrounds -- with the exception of the cosmic ray neutrons, to which the particular detectors used are not sensitive. To obtain the total background exposure it is necessary to add 11 mrem/yr to the TLD readings to allow for the neutron component (as taken from the dose-altitude curve of NCRP-45 at 2,200 m). The total exposures for the calendar year recorded by the TLDs at each station are listed in the annual reports of the Surveillance Group. Again from NCRP-45, the average exposure rate to cosmic radiation (excluding neutrons) at 2,200 m altitude is 60 mrad/yr. The average of the TLD readings for all seven stations over the six-year period from 1980 through 1985 is 116 mrad/yr. The average exposure from terrestrial radiation is, then, 56 mrad/yr.

Over any particular time period the cosmic background will, of course, be uniform across this compact array, though over a six-year period the level will change somewhat as a consequence of the 11-year solar activity cycle. At the geomagnetic latitude of the Continental U.S., this variation has a maximum amplitude of less than 10% of the mean level. Changes in the array average such as that between 1980 and 1981 (from 123 to 100 mrad/yr.), or that between 1982 and 1983 (from 109 to 131 mrad/yr.) will have resulted from changes in the terrestrial background. Presumably such shifts are to be accounted for by differences in precipitation, snow cover, and so forth -- and, indeed, there was 30% more precipitation in 1982 than in 1983: 21.7" vs. 16.7." However, the size of the changes from 1982 to 1983 was by no means the same at each station, ranging from +11 mrad/yr. to +35 mrad/yr. Another curious example of a station-to-station variation occurred between 1984 and 1985. The array average exposure was 116 mrad/yr. for each of these years; but, while the exposure at one station dropped from 135 to 120, that at another, only 1.2 miles away, increased from 115 to 136 mrad/yr.

The spread between the highest and lowest readings in 1980 was only 25 mrad/yr; but for each of the other annual periods this spread ranged between 30 and 40 mrad/yr -- even within the very limited extent of this array. During the six annual intervals considered, the lowest exposure was recorded at one or the other of two stations, while three different stations were involved in providing the highest reading of the year.

These examples, culled from the results of the Los Alamos survey, point up the fact that there is much more variability in the natural background radiation -- both over time, and in space -- than is brought to mind by references to countrywide, or even regional, averages.

(iv) Washington, DC

Being at sea level, Washington has a cosmic ray dose rate (including neutrons) close to 30 mrem/yr. Since the neutron component in this cosmic ray flux is quite small the difference between rads and rems is also small, and may be ignored. Washington is in the Coastal Plains Region for which the outdoor exposure rate to terrestrial radiation is said to be between 15 and 35 mrad/yr. The natural background dose rate in Washington should, then, be between 45 and 65 mrad/yr. Still, some question on this point is suggested by Alvin Weinberg's measurement in May 1979 of a dose rate of 250 mrem/yr during a hearing in the Dirksen Senate Office Building.

Having this in mind, a hand-portable radiation rate-meter was taken on several short excursions during May, June, and July of 1986. The resulting observations cannot be considered to constitute a survey, since they were made in the course of visits to a somewhat random selection of targets. The rapid time-response of the rate-meter made it attractive to take many of the readings en passant, so the precision of the readings was not impressive -- something like ±1 μR/hr. Still, the measurements were probably sufficiently accurate to permit the grouping into the rather broad exposure ranges indicated below. The rate-meter was calibrated in μR/hr; but that has been converted to mrad/yr using 1 μR/hr = 8.76 mR/yr = 7.6 mrad/yr.

The following is a summary of the results of this mini-survey. The numbers given refer to exposure rates in ambient air in mrad/yr.:

- 60-75. The lowest rate observed was about 60. This was found in a variety of locations: the doorway of the older World Bank Building at 18th and G; the 5th floor of the Hart Senate Office Building; at street level inside the new Presidential Plaza at 19th & I. Rates close to 75 were found along First Street, SE.; on the steps and among the columns in front of the Supreme Court; the northwest doorway of the Russell Senate Office Building; the interior of the Lincoln Memorial; and the street in front of 1717 H Street, as well as in the lobby and the large conference room on the 10th floor.

- 75-90. Examples were found along a number of streets (18th Street, I Street, Pennsylvania, and 20th); the lobby of the Lombardy Hotel; the

lobby of the National Science Foundation Building; and both on the street level and the lower level of the Farragut West Metro station.

- 90-115. Rates in this range were found on upper floors of both the Dirksen and the Russell Senate Office Buildings; on the street level of the new World Bank Building at 18th and Pennsylvania (then under construction) except that the rate of about 90 increased to about 115 on walking past the concrete structural columns; outside the base of the Washington Monument; the lobby of the Hay Adams; lobby of the New Executive Office Building; upper floors of the Lombardy Hotel; the men's rooms and corridors on the 10th and 11th floors of 1717 H Street (about 15 higher when passing concrete columns); the roadway of East Capitol near the foot of the steps to the Capitol; the sidewalk along Pennsylvania Avenue near the White House fence.

- 115-150. Inside the Washington Monument at ground level; beside the Reflecting Pool; in Lafayette Square (about 30 higher than on the other side of Pennsylvania Avenue); the street in front of the New Executive Office Building; on some sections of sidewalk such as that paved with bricks on Madison Place, and the section paved with ornamental stone slabs at 17th and H -- both being about 30 higher than nearby sections with concrete walks.

- 150-200. In this range were the entryway at the southeast corner of the Presidential Plaza; the porte-cochere on the east side of the Capitol; the walk by the Viet Nam Memorial; and the steps from the Reflecting Pool up to the Lincoln Memorial.

- >200. On crossing Madison Place from the east side of Lafayette Square (rate ~150) one can go through the porch of the Law Courts Building (rate ~265) into a delightful patio (rate ~240) and on into the lobby (rate ~120). On starting up the steps to the Library of Congress from First Street, SE. (rate ~75) one comes to the first landing (rate ~150), then the second landing (rate ~225), and then the doorway (rate ~380) and on into the lobby (rate ~115). On approaching the north entrance of the Old Executive Office Building one leaves the sidewalk on Pennsylvania Avenue (rate ~115), goes through a gateway in the fence (rate ~165), crosses a flagstone-paved patio (rate ~190) and up to the top of the steps (rate ~400) and into the lobby (rate ~135). Apart from these observations there is Weinberg's Senate Hearing room (rate ~250).

(v) Variability

Differences in natural background exposure rates of 50 to more than 100 mrad/yr have already been identified in earlier sections of this discussion. Such, for example, as that in the cosmic radiation background between locations at sea-level and locations at an altitude of 3 or 4 km., and as, also, that in the terrestrial background between the Coastal Plain and the Denver, Colorado Regions. Reference to these instances suggests broad, sweeping changes as between some location and another location a continent or part of a continent away.

However, the variations noted in the local surveys just described make it clear that the broad contours of the radiation intensity surface are overlaid by an irregular, fine-textured network of variations of appreciable size. It is not necessary for an individual to travel from the East Coast to Denver in order to encounter large changes in his rate of exposure to background radiation. Considerable variations will be experienced by a stationary individual in many locations, by individuals traveling a few miles to the store in many parts of the country (as evidenced by the NRC survey), by individuals residing in one house or in another house a few blocks away (as from the Los Alamos survey), or by individuals crossing from one side of the street to the other (as in Washington). Of course, the "country-wide, population-weighted, average annual exposure" is a perfectly well-defined concept which is useful for some purposes, even if there should not be an individual anywhere who actually receives just that exposure for one year, let alone two years running.

IV. Observations and Comments

We know that extreme exposures to radiation can be fatal, and we know a fair amount about the levels which produce lethal effects in a short time. We even know that there is some risk that an exposure about twenty times smaller than one resulting in a prompt fatality -- a whole-body exposure, that is, of something like 20 or 30 rem of low-LET radiation delivered in a short time -- may, with a rather poorly known probability, initiate processes which result in fatality years later. However, there is a gap of about two orders of magnitude between the dose levels for which observational data are available and the levels provided by natural sources. As stated in UNSCEAR-77, "It must be emphasized, however, that such estimates" (referring to their estimate of $\sim 10^{-4}$ fatal malignancies/person-rad) "are derived predominantly from rates observed following absorbed doses of over 100 rads," and "In particular, at low doses in the region of those received annually from natural sources, no direct information is available as to the level of induction of malignancies that might apply."

The human species has, of course, been receiving this natural background radiation, including variations of the sort already described, through the whole period of evolutionary time. Over that period it has evolved from primitive life forms, through the earliest hominids, and on to modern man. In the course of this it will have experienced a large number of mutations, of which some fraction will have been induced by natural background radiation. It seems to be generally felt that the outcome of this process has been favorable.

In its development the species has accommodated to the factors found in its environment; and for many of these factors there is a range of exposure levels which appear to be optimal for the well-being of the organism. Frequently this range is in the neighborhood of the levels usually encountered. Exposures (or supplies) at levels within this range may be either neutral, or beneficial, or essential to the organism's well-being; whereas great deficiencies may be detrimental or fatal, as may great excesses. Such is the case, for example, for the physical factors of heat, light, sound, and moisture. It is also the case for many chemical substances such as Vitamin A, and even materials containing arsenic and selenium. These, and many other substances, are essential in trace amounts but are deleterious or lethal at even moderate doses.

An agent having beneficial effects at low levels which would not be indicated by interpolation from its known deleterious effects at high levels is sometimes referred to as "hormetic." It is not known whether low-LET radiation is hormetic for the human organism, but it would not be greatly surprising if such should be the case. A large number of examples of radiation hormesis have been observed in a wide range of plant and animal species -- at least as gauged by such factors as growth rate, fertility, and longevity. Included in these observations is a series in which the rate of proliferation of a colony of bacteria increased as the radiation level was raised to fourteen times the natural background, but decreased both as the radiation level was raised still further, and as it was reduced by a factor of six below natural background by 10 cm of lead shielding. Suggestive as such observations may be, they are, of course, by no means conclusive as to an hormetic effect of radiation on the specific and complex system constituting human tissue.

In contrast with this there is no doubt that a single quantum can damage a cell or induce a mutation. A quite enormous number of experiments have been conducted on cell colonies and various types of animals showing deleterious effects of exposures of ~10 rads, or more. Such observations are also suggestive; but, as pointed out in UNSCEAR 77, they do not yet provide any direct information concerning the incidence of carcinogenesis in man resulting from exposures in the general range of natural background levels.

There are many cautionary statements by many authorities calling attention to the lack of actual knowledge on this last point. Those which appear in BEIR III include the following:

- "The Committee does not know whether dose rates of gamma or X-rays of about 100 mrad/yr are detrimental to man."
- "The quantitative estimation of the carcinogenic risk of low-dose, low-LET radiation is subject to numerous uncertainties. The greatest of these concerns the shape of the dose-response curve."
- "For the most part, the available human data fail to suggest any specific dose-response model."
- The collective influence of the uncertainties which apply "is such as to deny great credibility to any estimates that can now be made for low-dose, low-LET radiation."
- For its illustrative computations of the lifetime risk from whole-body exposure the Committee chose the situations of a single exposure to 10 rads, and a continuous lifetime exposure to 1 rad/yr, and then said: "Below these doses, the uncertainties of extrapolation of risk were believed by some members of the Committee to be too great to justify calculation."

In the face of these and other similar warnings that there is no factual basis for any particular estimate of the risk which might apply as a result of an exposure in the range of one to a few hundred mrad, precise values are routinely asserted for such quantities by regulators (and others) on the basis of the

no-threshold linear hypothesis. Of course, different precise values are equally firmly asserted by different estimators since the slope of the line employed is open to some choice. For example, the average risk of inducing a fatal malignancy was taken as being "in the region of 10^{-4}/rad" in UNSCEAR-77; but the coefficient multiplying 10^{-4} in the estimate of the number of fatalities per man-rem has been variously taken to be: 1.25 (ICRP-26, 1977), 1.7 (BEIR 1972, 1980, and 10 CFR 20, NRC, 1985), 2.3 (NUREG-1150, 1987), 2.7 (40 CFR 191, EPA, 1985), and 3.75 (40 CFR 193, EPA, 1987).

Though these assorted values display the lack of any absolute technical basis for the assumptions, to a very large extent the no-threshold linear hypothesis has been accorded the status of an axiom. Two important corollaries follow from this hypothesis. The first is that the risk to the individual is directly proportional to the dose (that is, the increment in exposure over natural background plus radiation received for medical purposes) independently of the size of the incremental exposure or the level of the background. (The medical component, incidentally, though highly variable from person to person, is estimated (NCRP, 1987) to add 53 mrem/yr to the average exposure of the population of the U.S.). The second corollary is that the collective consequences of an incremental exposure of a population are directly proportional to the integral of the incremental number of man-rem delivered, independently of the size of the incremental exposures. For hypothetical incidents, at least, the incremental total of man-rem is itself the endpoint of a chain of assumptions concerning meteorological factors, individual behavior, and so forth.

Not everyone subscribes to the linear hypothesis. There are those who hold that at low exposures the dose-response curve is concave downwards and lies above a straight line from the origin so that the effects at low doses will be larger -- possibly much larger -- than indicated by the linear hypothesis. There are also those -- including the majority of the BEIR III Committee -- who consider it probable that the true response curve is concave upwards, lies below a straight line through the origin, and that the effects at low doses will be less -- possibly considerably less -- than indicated by the linear hypothesis. And then, as mentioned above, there are those who hold that radiation may be hormetic. For those, exposures in some range of low doses would not necessarily constitute any risk at all. Arguments can be (and have been) adduced in support of each of these dose-response models; but, as already noted, the BEIR III Committee concluded "the available human data fail to suggest any specific dose-response model."

Whether the fashionable linear hypothesis represents any biological reality or not, it does have two features in its favor. One (for which few public claims are made) is that it is wonderfully economical of regulatory thought. The other (that most commonly urged in its defense) is that it is said to be "prudent." This neo-technical term -- like its sister regulatory term "conservative" -- is frequently invoked to provide an unassailable license to make mistakes, as long as they are made in the right direction, i.e., to overstate the negative aspects. However, other considerations must also be taken into account, and a "prudent" estimate on one side of an equation is unlikely to be of assistance in striking a (truly) prudent balance between conflicting considerations -- a balance, that is, reflecting the exercise of good judgment.

The ALARA principle may be taken as an example. Here the intention is to reduce risk, but the quantity which can be directly affected by actions taken is potential radiation exposure. It becomes progressively harder (and more expensive) to reduce the potential exposure the lower the level at the start. The resources devoted to the exercise of ALARA are themselves of interest to the public; but the ALARA principle is open-ended. To avoid the indefinite and ultimately pointless iteration of cost-benefit analyses along with efforts to devise means of reducing potential exposures to ever lower levels, there is an obvious need for some sort of floor for the further imposition of ALARA. Of course, along with ALARA and the linear hypothesis we do have the $1,000 per man-rem convention; but this merely intercalibrates the scales for risk estimates and costs, albeit in a somewhat arbitrary fashion. It enables cost-benefit analysis, and may serve to support a decision that some particular mechanical measure estimated to reduce exposures by such and such an amount is not "worthwhile." But it goes no way towards saying when further study may be laid aside. Neither the linear hypothesis -- and particularly not the presumed prudence of the risk estimates derived from it -- nor the $1,000 per man-rem convention provide any logical assistance in specifying a reasonable floor for ALARA.

Similar conditions apply to attempts to establish a de minimis dose. While this and a floor for ALARA have much in common they are by no means identical. The one refers to a level at which further efforts at reduction would not be mandatory. The other refers to a dose level at which the consequences (if any) would be deemed trifling and would not warrant consideration -- either by regulators, individuals, or society. It has been urged that the establishment of a de minimis level would serve a number of useful purposes, such as providing a cut-off for regulatory efforts (presumably including estimation of collective doses), providing limits for control programs, and, conceivably, assisting in developing a better public understanding of the significance of radiation exposure. Any level selected will have to meet a number of conditions, among them that it can be adequately measured, but also that it be capable of gaining public acceptance.

The desirability of an official de minimis dose has been discussed for many years among many groups. Most frequently such discussions have started (and often ended) with attempts to decide on an "acceptable risk." Levels for the risk of premature death in the range of 10^{-8} to 10^{-5}/yr have been mentioned in this connection and, with the help of the linear hypothesis, a corresponding range of dose levels: from less than one to over a hundred mrem/yr. It is, of course, far from clear what an "acceptable" risk level may be, or even if there is one. Much has been made of the fact that even in so-called "safe" industrial settings the (occupational) risk may run as high as 10^{-4}/yr; and, since this appears to be acceptable, presumably any risk appreciably smaller -- such as 10^{-6}, say -- ought to be acceptable too? However, this may not cover the situation. For one thing, the risks in familiar settings have not been so flamboyantly identified, debated, and belabored as they have for radiation, and may to a large extent be accepted unknowingly. For another, there is nothing to say that risks similar to or even smaller than those applying to more familiar activities would be deemed acceptable for radiation -- partly in view of its being pictured as more mysterious, but at least partly because the official assumptions have tended to endorse an unlimited and unreasoning fear. Such psychological factors could well interpose great difficulties for any risk-based

approach to settling on a reasonable prescription for a de minimis dose. There is also the more basic point that to proceed from a pre-assigned risk to an associated dose in a logical way one really needs a fairly well-founded dose-response correlation; but, as BEIR III has tirelessly told us, we do not have one.

V. Conclusion

Confronted with a zone of ignorance a few decades wide we have adopted the simplest possible hypothesis for use as a convenient bridge. A priori it would seem rather unlikely that a linear dose-response function would actually provide a very good description of such a complex biological relationship as that between carcinogenesis and radiation exposure. Perhaps the best that can be said of it is that there is a majority opinion that it provides a "prudent" description. This cannot be regarded as an intellectually satisfactory basis for important decisions, particularly when there are some quite relevant facts which are known with certainty. Amongst these is the wide variation in the exposures people receive naturally.

As a sufficient gesture towards "prudence" one can leave aside the most extreme situations -- such as the Andes, the Kerala Coast, or even the hearing rooms in the Dirksen Senate Office Building -- and still find that millions of people dwell in low-LET radiation fields with levels from 50 to more than 100 mrem/yr larger than the averages usually assigned to the natural background of low-LET radiation. There is no evidence that the variations encountered by these quite large contingents of the species are detrimental in any way. Such variations are similar in nature to the other inhomogeneities which mark the planet we inhabit: differences in weather and climate, sunlight, altitude, water, ice, and so forth. They are features of the environment in which we have developed, and in which we will continue to live. For the particular factor of exposure to radiation the natural background, and its variations, provide the most certain guide and basis we have (and, quite possibly, the most certain guide we ever may have) for consideration of such matters as appropriate levels for a floor to ALARA and for a de minimis dose. (These, of course, need not be the same.) Though the guidance which might be drawn from observation of the broad spectrum of natural variations would have a real basis, that would not point clearly at any precise value that should be used for the purposes we are considering. It is not proposed to try to specify such a value here. However, from the evidence which has been presented it would appear that any attempt to argue for a smaller level than 50 or 100 mrem/yr would have to construct its support on the mystical basis of the linear hypothesis.

The main intent of this discussion is to urge that we base our actions and decisions to the extent possible on things known. Quantitative statements concerning risk do not fall in that category. Particularly offensive in this respect are the statements frequently emitted by regulators to the effect that so and so many "excess fatalities" will result from such and such an incremental exposure, or be saved by this or that proposed new procedure. Such statements are not only without foundation -- derived, as they are, by the application of some simple scale factor of unknown validity -- but they also tend to give the unwarranted and revolting impression that fatalities are the coin of the realm in nuclear affairs. It is really of great importance that comments concerning risks, fatalities, and so forth should be as factual as possible. They should leave no room for doubt as to what the real situation is, namely: that neither we, nor anyone else, knows the precise relationship

between dose and response at low doses; but that, at least within the range of exposures discussed above, there is as yet no evidence of any detrimental effects on man.

NOTE CONCERNING RADIATION UNITS

The units used throughout this discussion are the "rad" and the "rem."

The rad is the unit for energy deposited by ionizing radiation of any type in any material. One rad refers to an exposure resulting in an absorbed energy (or "dose") of 100 ergs/gm. To provide a correlation between the radiation flux and the dose, it is necessary to specify the material considered. Thus, one speaks of "an absorbed dose in air" of so many rads. Since there are more electrons per gram in biological tissue than in air (resulting from the larger proportion of hydrogen in tissue) a given flux of radiation will deposit somewhat more energy per gram in tissue than in air. This difference, however, is rather small (only 7%) and is usually ignored; so that, to a reasonable approximation, a radiation exposure providing an absorbed dose in air of one rad would be said to provide one rad of absorbed dose in tissue.

The rem is the unit used to calibrate biological effects in human tissue. One rem is the dose from any radiation that produces biological effects in the body equivalent to those from one rad of X-rays within a given energy range. One rad delivered by α-particles is more damaging then one rad from X-rays, even though the amount of ionization per gram produced by the two would be the same. In the case of the α-particle most of its energy is deposited in a very short distance at the end of its path, resulting in a very high level of ionization within a quite small volume. On this account an α-particle is said to have a high Linear Energy Transfer (LET), and the radiation it provides is referred to as high-LET radiation. By contrast, an X-ray deposits its energy more uniformly along the length of its (longer) path, and X-rays (as well as γ-rays and β-particles) provide what is called low-LET radiation. To take account of the differences in biological effects, a factor -- variously referred to as the Relative Biological Effectiveness (RBE) or Quality Factor -- and, in current writing, usually designated Q -- is introduced, by which the dose in rads is multiplied to obtain the dose in rems. By definition, Q=1 for X-rays; and is also taken to be unity for γ-rays and β-particles. However, for α-particles, it is now officially agreed to take Q=20. A value of Q between 5 and 20 is currently assigned to neutrons.

One further convention is necessary. α-particles as well as β-particles, deposit their energy in a quite thin layer of tissue immediately adjacent to their source. Thus, they do not provide a "whole-body" dose, but only a dose to the organ (or small portion thereof) in which the source of such radioactivity may be located. In order to assess the relative biological hazards of the effects of radiation delivered by various means to various parts of the body, one needs some way of translating any particular organ dose onto a common scale at a level judged to represent an equivalent overall effect. The ICRP has devoted extensive efforts to developing a system of "weighting factors" whereby the dose to any particular region of the body can be converted to an appropriate value of whole-body dose equivalent (D.E.). This is called the "effective" D.E. Thus, for example, the effects of a dose of x rem to the lung tissue is taken to be adequately represented by 0.12 x rem of whole-body D.E. Obviously, the sum of all the weighting factors for the different organs, or

body judged to be significant, must equal unity -- so that a
each significant center will, when added up, equate to a
of xrem.

comment, it may be noted that the classical unit for radiation
Roentgen (R) -- is no longer in use, though it appears in many
cent reports. The Roentgen was defined in terms of the amount
duced; in particular: one esu of charge in one cubic centimeter
... at standard temperature and pressure. This is equivalent to an absorbed dose in air of 87 ergs/gm (0.87 rad), or of 93 ergs/gm in tissue. At least in discussions of natural background radiation (where air and tissue are the media of interest) the rough approximation 1 R = 1 rad is frequently used. Today, there is the newer SI unit for energy deposited -- the Gray (Gy). One Gray is the exposure resulting in the deposition of one joule/kilogram (rather than 100 ergs/gm) so that 1 Gy = 100 rads. Similarly, the SI unit for dose equivalent in biological tissue -- the Sievert (Sv) -- is such that 1 Sv = 0 Gy = 100 rems.

Finally, just as one could (if one chose) calibrate velocity in terms of furlongs per fortnight, there is the unit of the Working Level (WL) to calibrate α-activity in air, and the Working Level Month (WLM) for the integrated exposure to such radioactivity. The WL is defined as a concentration of short-lived radon daughters which would result in the release of 1.3×10^5 Mev of α-energy per liter of air. The population of the radon daughters Po-218, Pb-214, Bi-214, and Po-214 in radioactive equilibrium with 100pCi/l of Rn-222 would release 1.3×10^5 Mev of α-energy per liter, and would thus provide one WL. The WLM is defined as the exposure to one WL for 170 hours. As with any attempt to correlate the concentration of airborne radioactivity with the dose delivered to any particular organ (such as the lung) resulting from inhalation, the steps are more than a little complicated; requiring, as they do, either knowledge or assumptions concerning breathing rate, departures from radioactive equilibrium (which essentially always apply except in situations where the air is quite stagnant), the particle sizes of the aerosols involved, and the extent to which the individual radon decay products are attached (or not attached) to the dust particles within the air, as well as the physiological distribution and retention of the materials inhaled. On the basis of averaging assumptions on each of these points it has been taken that one WLM corresponds to a dose of about 12-14 rem to the segmental bronchioles. With the ICRP weighting factor of 0.08, one WLM corresponds to a whole-body dose equivalent of about 1 rem.

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RADIOACTIVITY RELEASED FROM PHOSPHATE-CONTAINING FERTILIZERS AND FROM GYPSUM

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Abstract—Large amounts of uranium, radium and radium decay products are redistributed throughout the environment due to the use of phosphate fertilizers. Potential radiological impacts resulting from direct exposure, surface run-off, inhalation and ingestion of foods grown with fertilizers are discussed. Also, waste gypsum piles, a by-product of fertilizer production, are a significant source of radon emissions to air.

INTRODUCTION

Fertilizers containing phosphate have become essential to the world's agriculture. They are produced and used worldwide in increasing quantities to replenish natural nutrients depleted from soils because of farming and erosion. However, the mining, processing and use of fertilizer materials in massive quantities redistributes radioactive trace elements throughout the environment, principally uranium, radium and radium decay products.

Gypsum is the by-product of the production of phosphoric acid used in the manufacture of phosphate fertilizers. Phosphate rock is treated with sulfuric acid to produce phosphoric acid, which is then combined with either ammonia or a form of marketable phosphate rock to produce ammonium phosphates and concentrated superphosphates, respectively. A form of impure calcium sulfate called gypsum is removed as a precipitate. This gypsum is normally disposed of simply by placing it into large tailings piles, which remain indefinitely. Most of the radium in the phosphate rock is co-precipitated with the gypsum. Thus, gypsum piles become a source of radioactivity released into the environment.

THE MANUFACTURE OF FERTILIZERS

Fertilizers are made up of materials containing nitrogen, phosphorus and potassium. Generally, the nitrogen is derived from an ammonia based material, the phosphorus is obtained from phosphate rock, and the potassium comes from potash. The nitrogen portion contains negligible radioactivity. The phosphorus portion may contain substantial concentrations of uranium, thorium and radium and their decay products. The potassium portion has a small amount of potassium-40, in accordance with its general concentration in nature.

Drastic chemical treatment with strong acids (sulfuric or phosphoric) is necessary to produce soluble phosphate products. This technique of producing primary fertilizer products is called wet process phosphoric acid. In this process, phosphate rock is combined with sulfuric acid to form normal superphosphate. This fertilizer product is a mixture of phosphoric acid and gypsum (calcium sulfate). If the gypsum is filtered out of the material, the resultant liquid fertilizer product is phosphoric acid. By combining phosphoric acid with phosphate rock, another fertilizer product, triple superphosphate, is produced. Finally, combining phosphoric acid with ammonia forms various ammoniated phosphate fertilizers. These processes and their products are illustrated in Fig. 1.

SOURCES OF RADIOACTIVITY IN FERTILIZERS AND GYPSUM

The radiological importance of fertilizers of a specific country is dependent on where the phosphate rock used to make the fertilizer is mined. Investiga-

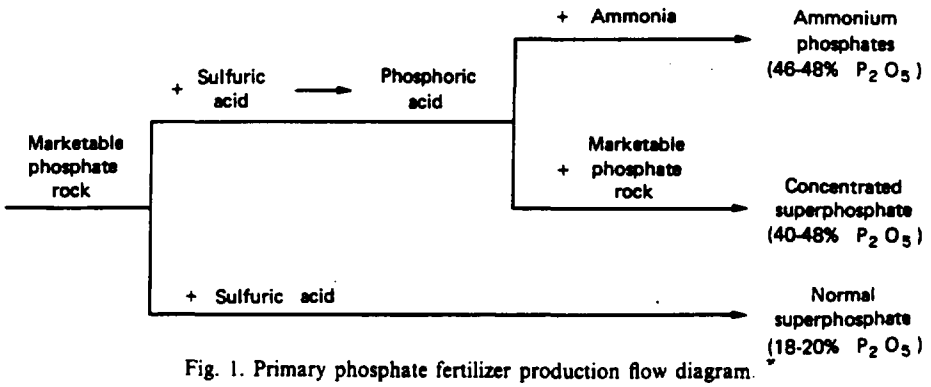

Fig. 1. Primary phosphate fertilizer production flow diagram.

Table 1. Radioactivity (Bq/g) in phosphate rocks

Rock origin	Percent phosphate	Ra-226	U-238	Th-232	K-40
Morocco	34	1.6	1.7	0.01	0.02
Morocco	35	1.6	1.7	0.02	0.01
Taiba-Togo (calcined)	35	1.1	1.3	0.03	0.004
Bu-Craa (Western Sahara)	34	0.9	0.9	0.007	0.03
Kola (U.S.S.R.)	39	0.03	0.04	0.08	0.04
U.S.A. (Florida)	32	1.6	1.5	0.02	
U.S.A. (Western)	31	1.0	1.0		0.02

tors have reported a wide variation in the concentrations of radionuclides in phosphate rocks (Menzel, 1968; Guimond and Windham, 1975; Pfister *et al.*, 1976). Typical concentrations of uranium and radium in phosphate rock from various places in the world are listed in Table 1. Uranium is reported to range from about 0.1 to 10 Bq/g, but there can be substantial variations in the concentrations of a given radionuclide within a specific country. For example, in the United States, radium concentrations range from about 0.2 to 2 Bq/g, with ores mined in Florida having the highest concentrations and ores mined in Tennessee having the lowest.

In 1983, about 135 million metric tons of phosphate rock were produced worldwide (Stowasser, 1985). About 46% originated in the United States or Morocco, and the remainder came from over 25 different countries in North America, South America, Europe, Africa, Asia and Oceania. These production data are summarized in Table 2. International trade in phosphate rock is summarized in Table 3.

There are very few analytical data available regarding the radioactivity content of the multitude of fertilizer mixtures used in the world. The concentrations of uranium, thorium, and radium and their decay products in some commercial fertilizers in the United States have been studied by the U.S. Environmental Protection Agency (Guimond and Windham, 1975; U.S. Environmental Protection Agency, 1977, 1978). Also, the concentrations of radionuclides for various fertilizer products used in Germany were measured by Pfister *et al.* (1976). These data are listed in Tables 4 and 5, respectively.

The differences in the radioactivity concentrations of the various basic fertilizers listed in Table 4 are principally due to the partitioning that occurs during the acidulation phase of phosphoric acid production (Guimond and Windham, 1975). Most of the radium stays with the gypsum, whereas the uranium and thorium remain with the phosphoric acid. Since normal superphosphate contains both phosphoric acid and gypsum, it has the largest concentration of radionuclides. Commercial fertilizer blends made from normal superphosphate would be expected to contain more radium and other radionuclides than fertilizer blends made from other basic fertilizer materials. Commercial fertilizer products derived from phosphoric acid would be expected to contain the smallest concentrations of radium, but they may have substantial concentrations of uranium. However, in the United States, many plants have installed

Table 2. World phosphate rock production—1983

Country	Production (kt)	Country	Production (kt)
North America		*Asia*	
U.S.A	42,573	China	12,669
Mexico	700	Christmas Is.	1,094
		Israel	2,969
South America	3,229	Jordan	4,749
		Vietnam	220
		Other	3,614
Europe			
U.S.S.R.	27,200		
Other	498	*Oceania*	
		Australia	21
		Nauru	1,684
Africa			
Algeria	898		
Morocco	20,106		
Senegal	1,249		
Rep. of S. Afr.	2,742	*Total*	135,000
Togo	2,081		
Tunisia	5,924		
Other	780		

Table 3. International trade in phosphate rock—1983 (principal exporting countries)

Exporting country	Destination	Quantity (kt)
U.S.A.	Canada	2648
	Western Europe	3776
	Asia	3498
	Eastern Europe	863
	South America	440
	Oceania	390
Morocco	Western Europe	9445
	Eastern Europe	2760
	South America	802
	Asia	1385
	Oceania	35
Algeria and Tunisia	Western Europe	722
	Eastern Europe	846
	Asia	20
Israel and Jordan	Western Europe	2148
	Asia	1694
	Eastern Europe	1454
	Oceania	80
Senegal	Western Europe	877
	Asia	315
	Eastern Europe	57
Togo	Western Europe	1174
	Eastern Europe	799
	Asia	20
U.S.S.R.	Eastern Europe	3701
	Western Europe	1193
Pacific Islands	Australia	1687
	New Zealand	756
	Others	116

Table 4. Radioactivity (Bq/g) in United States fertilizer products

Product	Percent phosphate	Ra-226	U-238	Th-232
Superphosphate	18	0.78	0.74	
Triple Superphos (Florida)	38	0.78	2.1	0.049
Triple Superphos (Western)	38	0.52	1.6	0.17
Phosphoric Acid (Florida)	28	0.037	0.93	0.11
Monoammonium Phos (Florida)	54	0.19	2.0	0.06
Monoammonium Phos (Western)	54	0.031	1.0	0.074
Diammonium Phosphate (Florida)	46	0.21	2.0	0.06
Diammonium Phosphate (Western)	46	0.024	0.78	0.003

Table 5. Radioactivity (Bq/g) in German fertilizer products

Product	Percent phosphate	Ra-226	U-238	Th-232
Superphosphate	18	0.52	0.52	0.015
Triple Superphosphate	38	0.23	0.80	0.044
Thomasphosphate	14–17	0.007	<0.04	<0.004
Hyperphosphate	29	0.84	0.88	0.16
Novaphosphate	23	0.75	0.78	0.015
Carolonphosphate	26	0.57	0.92	0.03
Rhenania-Phosphate	28–30	0.03	<0.04	0.06
Kalciphosphate	38–40	0.055	0.81	<0.004
ENPEKA-Ammon-phosphate	52	0.089	2.3	0.048
Diammonium Phosphate	45	0.019	2.2	0.019

Table 6. Radioactivity (Bq/g) in selected German fertilizer blends

Commercial name	Percent phosphate	Ra-226	U-238	Th-232
Complesa (15/15/15)	15	0.34	0.54	0.011
ENPEKA (12/12/17)	12	0.29	0.67	0.004
KAMPKA (13/13/21)	13	0.055	0.25	0.022
Neues AmSupka	12	0.50	0.63	0.007
NITROPHOSKA (10/15/20)	15	0.26	0.62	0.019
RUSTICA (13/13/21)	13	0.007	<0.04	0.044

solvent extraction circuits, enabling the recovery of uranium as a co-product rather than shipping it as a contaminant in the fertilizer. Five plants in the United States now have the potential to recover up to a total of 1.7 million kg of uranium oxide each year (Stowasser, 1985a).

Commercial fertilizer blends made from triple superphosphate, ammonium phosphates, or other basic fertilizer materials are likely to have radium and uranim concentrations lower than the primary phosphate fertilizers. Typical concentrations are expected to be about 10–50% of the concentrations listed in Tables 4 and 5, because the basic fertilizer products are used in those proportions in many fertilizer blends. This is demonstrated by the data in Table 6 as obtained by Pfister for some blended and mixed fertilizer products used in Germany (Pfister *et al.*, 1976).

RELEASE OF RADIOACTIVITY FROM FERTILIZERS

The release of uranium and radium to the environment due to fertilizer use may be illustrated using United States fertilizer consumption data. Weighting United States consumption data by proportions originating in the South and West of the country and applying the appropriate concentration data from Table 4 (Stowasser, 1985b) produces the estimates of 14,000 and 990 GBq of uranium and radium respectively entering the United States biosphere each year, as shown in Table 7. This is only a small percentage of the total radium associated with fertilizer production as most of the radium remains in the gypsum waste piles. As fertilizer production moves towards phosphoric acid and ammoniated phosphates rather than normal superphosphate or triple superphosphate, there is more efficient removal of the radium from the wet process plant to the gypsum pile.

Using the same method suggests that about 4000 GBq of radium are introduced into Western

Table 7. Radioactivity (GBq) in fertilizer used in the United States (1984)[a]

Region	Phosphate (kt)	Ra-226	U-238	Th-232
Northeast	220	48	710	19
Lake states	540	120	1,700	45
Corn belt	1500	330	4,800	130
Northern plains	460	100	1,500	39
Appalachia	400	88	1,300	34
Southeast	320	70	1,000	27
Delta states	130	42	610	16
Southern plains	350	77	1,100	29
Mountain states	240	53	770	20
Pacific	270	59	870	22
Total	4490	987	14,360	420

[a] Estimates of total radioactivity developed from data on fertilizer production in the United States for 1984 by weighting the production data by proportions originating in the South and West and applying concentration data from Table 4.

Europe each year in fertilizers (Stowasser, 1985a). This is likely to be an underestimate because it is based only on the phosphate rock imported into Western Europe and does not include the fertilizer products directly imported.

Release through run-off and leaching

Elements such as phosphorus and potassium are not generally recycled to the biosphere in large quantities. These elements, if not intercepted by plants or held by soils, have a one-way journey to the sea. This is also the fate of the various radionuclides that are incorporated with potassium and phosphate nutrients (Miller, 1972; Mistry *et al.*, 1970).

Spalding examined uranium concentrations in numerous rivers flowing into the Gulf of Mexico (Spalding and Sackett, 1972). His studies indicated increased uranium concentrations when compared to 20-year-old data for the same rivers. He attributed these increases to the widespread application of phosphate fertilizers in agriculture. Radium is likely to be more insoluble and to stay with the sediments. Increased recovery of uranium from phosphoric acid should reduce the amount of uranium available to the environment as run-off in future years.

Comparison to Table 7 shows that about 40% of the total radioactivity distributed by fertilizer use in the United States during 1984 was in the Mississippi basin States. A rough estimate suggests that about 390 GBq of radium and 5700 GBq of uranium were present in the fertilizers used in Illinois, Iowa, Minnesota, Missouri, Wisconsin, Kentucky, Tennessee, Mississippi, Arkansas and Louisiana. Consequently, the Mississippi and its water basin probably receive the greatest amount of radioactivity from agricultural run-off in the United States.

Release through crop uptake

Since most fertilizers are used to improve crop yields, there is concern about the potential uptake by plants of the trace radionuclides present in the fertilizers. Using mean values of the specific activities of natural radionuclides in phosphate fertilizers, Pfister estimated the maximum build-up of uranium and radium in soils that might occur in Germany due to fertilizer usage (Pfister *et al.*, 1976). A maximum fertilizing intensity of 1525 kg/ha was reported in the region of Wurzburg in 1973–74. From this, he calculated that about 3.2 MBq of uranium and 2.2 MBq of radium might be added to the soil each year. Much of the uranium, radium and thorium is likely to be strongly adsorbed by the soil, as is phosphate (Shultz, 1965). Consequently, there is some build-up of radionuclides likely to occur in heavily fertilized land. Guimond and Pfister estimated that such build-up may range from a few percent to greater than the original radionculide concentrations in the soil (Guimond, 1978; Pfister *et al.*, 1976). Radium and uranium are normally present in soil in concentrations of about 0.004 to 0.11 Bq/g of soil.

Studies by several investigators have indicated that food crops take up radionuclides such as radium and uranium from the soils in which they are grown (Mistry *et al.*, 1970; Penna Franca *et al.*, 1965; Watters and Hansen, 1979). The amount of uptake has been shown to be dependent upon several factors, including solubility, crop type, soil type, and calcium concentration in the soil. In most cases, the relative concentration factors (concentration in dry plant material/concentration in dry soil) was less than 0.1, suggesting that the radionuclides are excluded to varying degrees. None the less, increases in soil radioactivity concentrations due to fertilizer use may be accompanied by some increases in the radioactivity present in various food crops, although generally it should be small. Investigations in New Zealand, Australia and the United Kingdom have shown increases in the respective α-activity of wheat grains and sheep bones after wheat fields and grazing areas were intensively fertilized with superphosphate (Marsden, 1964).

Releases to air and exposure to γ-radiation

Windham studied workers in wet-process fertilizer plants in the United States (Windham *et al.*, 1976). He reported that workers who come in close contact with large amounts of phosphate ore and fertilizer products are subject to the inhalation of dust generated by unloading, crushing, drying and transport. He reported a maximum potential dose equivalent to the lungs of about 50 mSv/y. He also estimated that direct γ-dose equivalents for workers ranged from about 0.3 to 3 mSv/y. There are no data on workers at blending and mixing plants and storage facilities, but direct exposures from worker proximity to large quantities of fertilizer could be similar to the direct radiation exposures measured at wet-process phosphoric acid plants. Such exposures could substantially exceed typical natural radiation background exposure.

Umwelt calculated that, based on fertilizer use in Germany, an external radiation exposure of 1.1 μSv/y could occur to the gonads and bone marrow from one application of phosphate fertilizer. By estimating the total amount of fertilizer applied during an 80-year period, he calculated that a member of the general public could receive about 17 μSv/y from external exposure to phosphate fertilizers and that agriculturally employed persons could receive about 20 μSv/y (Umwelt, 1975). Pfister made similar calculations based on slightly different assumptions. He concluded that a member of the general public was not likely to receive more than about 4 μSv/y due to fertilizers and that an agricultural worker would receive about 13 μSv/y (Pfister *et al.*, 1976).

RELEASE OF RADIOACTIVITY FROM GYPSUM PILES

There is a large amount of by-product gypsum now stored in waste tailings piles. There are approximately

60 gypsum piles at 40 different locations in the United States, primarily in the States of Florida, Idaho, Illinois, Louisiana and Texas. These piles are a by-product of an industry that produces phosphoric acid at a rate that reached approximately 10 million tons in 1981 (The Fertilizer Institute, 1982). The total surface area of these piles is about 3300 ha (U.S. Environmental Protection Agency, 1987). As the worldwide consumption of phosphate is approximately five times that of the United States, the world-wide gypsum pile inventory may be about in the same proportion (The Fertilizer Institute, 1982).

In the United States, the gypsum is usually transferred to nearby disposal areas with little or no prior preparation of the land surface. The gypsum slurry is pumped to the top of the pile, where it forms a small impoundment. The pile is enlarged by dredging gypsum from the impoundment to increase the height of the walls that enclose the impoundment. Such piles may eventually reach 300 ha in area and 60 m in height (U.S. Environmental Protection Agency, 1987). Most of the radium in the original phosphate rock processed through phosphoric acid plants remains in the gypsum piles.

Radioactivity in the gypsum pile may escape by a variety of mechanisms. The radium becomes a source of radon, which continuously diffuses to the surface of the pile and escapes to the air. When the pile dries out there may be wind erosion at those locations where the surface is disturbed, e.g. by a bulldozer or other vehicles. Then, radioactive particulate matter may escape from the pile into the air. Rain water may pass through the pile, leaching radionuclides into the ground water and nearby surface waters. Finally, if this by-product gypsum is removed from the pile and used as a building material or soil conditioner, or for some other use, there is usually a greater probability that radionuclides will escape into the environment.

Releases to air

Horton has measured the radon emission rates for two phosphate gypsum piles in central Florida (Horton, 1986). Both piles had an average radium concentration of about 0.9 Bq/g. The release rate of radon was measured to be about 3600 Bq/m^2/h; the total annual release rate was estimated to be 25 TBq/y for a pile with an area of 82 ha. The escape rate varied by almost two orders of magnitude at various measured points on the pile. This variation can be explained by nonuniform distribution of radium in the pile material and moisture content in the pile. Meteorological parameters such as barometric pressure, wind speed, ambient temperature, and humidity might potentially influence the release rate.

Hartley made more detailed measurements of radon emission rates for two phosphate gypsum piles in Florida over a 4-day period (Hartley and Freeman, 1986). The average radon escape rate was 2500 Bq/m^2/h. For the drier areas, the escape rate was 1500–3800 Bq/m^2/h; while for the wet areas it was an average factor of 9 less. Locations on the pile covered with 8–15 cm of soil had an average radon escape rate of about 900 Bq/m^2/h.

Table 8. Radionuclide concentrations (μBq/M^3) in particulates collected near a gypsum pile

Radionuclide	460 m upwind from pile edge	115 m downwind from pile edge
U-238	3.7	5.6
U-234	4.1	5.9
Th-232	4.1	5.6
Ra-226	4.1	7.0

The U.S. Environmental Protection Agency has recently measured radon releases at four active and one inactive gypsum piles over a year-long period (U.S. Environmental Protection Agency, 1987). Measurements were made at weekly and monthly intervals at 10 sites on each pile. The overall escape rate for all locations on the four active piles was 2500 Bq/m^2/h; for the inactive pile for the same period, the escape rate was 500 Bq/m^2/h. Experience in central Florida has shown that when gypsum piles become inactive, a thick crust forms, which acts a natural barrier to radon emissions, reducing the escape rate by about a factor of five. The concentration of radium in these piles acting as the source of the radon was approximately 1.1 Bq/g. The corresponding uranium concentration was about 0.1 Bq/g.

The U.S. Environmental Protection Agency (1987) has estimated the total release of radon from all gypsum piles in the United States as being about 330 TBq/y from a total surface area of 3300 ha.

When there is vehicle traffic on the pile, particulate emissions to air from gypsum piles can be observed. Wind erosion normally does not occur because of the protection afforded by the high moisture content of the active piles or the crust that forms on inactive piles. Airborne radioactive particulates have been collected in the vicinity of a gypsum pile over a 4–6 month period. Upwind and downwind radionuclide concentrations are shown in Table 8 (U.S. Environmental Protection Agency, 1987). However, not all of this radioactivity was thought to be due to the pile; most may be due to dust from the ground. The annual particulate escape rate was estimated for a typical pile by means of a fugitive dust emissions model (U.S. Environmental Protection Agency, 1987). These estimates are shown in Table 9 and

Table 9. Estimated emissions in particulates released from a typical gypsum pile due to vehicular traffic[a]

Radionuclide	Emission rate (MBq/y)
U-238	2.3
U-234	2.4
Th-232	3.7
Ra-226	23
Rn-222	23
Pb-214	23
Bi-214	23
Pb-210	26
Pb-210	20

[a]A generic pile of 31 ha.

represent particulates that are 30 μ or less in dust escaping from the pile due to 4000 km of traffic movement per year. Most of the risk due to inhaling these particulates will be due to the radium emissions of 23 MBq/y.

Release to ground and surface waters

The radium in gypsum piles is most likely in the form of radium sulfate, an extremely insoluble compound. Therefore, it might be expected that significant amounts of radium would not leach from the pile into ground and surface waters. Strain has measured the radium and radon concentrations in ground and surface water near a phosphate mining and manufacturing facility in North Carolina, finding some evidence of release of radium to these waters (Strain, 1979). Most well water samples in the area around the facility had radium concentrations of less than 0.07 Bq/l; radon concentrations were typically less than 20 Bq/l. Significantly higher concentrations of radium and radon, as much as 6 Bq/l and 700 Bq/l, respectively, were observed in four wells. Concentrations of radium in river water downstream of the plant were twice as high (0.1 Bq/l) as in water upstream of the plant (0.05 Bq/l).

In the Netherlands, 2 Tg/y of phosphogypsum is disposed of into the Rhine River. Koster has noted this has led to an increase of uranium-238 chain radionuclides along the Dutch coast off Rotterdam, decreasing in a northern direction into the German bight (Koster *et al.*, 1985).

Releases due to use as building materials

Gypsum is used in building when it is processed into wallboard for use as an inner wall construction material. Natural gypsum low in radium is normally used for this purpose, but the potential for using by-product gypsum has lead to investigations of the radon escape rate from this material. Mustonen has measured rates of radon emission from by-product gypsum building material with different thicknesses and radium concentrations. Values are given in Table 10 for Finnish material (Mustonen, 1984). As example, for a 10-cm thick slab of gypsum with a radium concentration of about 0.5 Bq/g, the escape rate measured was 20 Bq/m^2/h. The rate of escape of radon from gypsum is considerably less than that from concrete slabs of the same thickness, about a factor of five. But due to a lower radium content in Finnish concrete, the overall radon escape rate from concrete is about a factor of two lower than for by-product gypsum.

Table 10. Radon released from gypsum slabs

Gypsum slab thickness (cm)	Radium concentration (Bq kg^{-1})	Rate of radon release (Bq m^{-2} h^{-1})
6.7	320	14
20	480	42
15	480	31
10	480	20
7.5	480	19
5.0	480	10
2.5	480	4.7

Releases due to use as soil conditioners

The large amounts of phosphogypsum available have motivated searches to find useful applications, or at least cheaper disposal methods. As a part of this effort, phosphogypsum has been mixed with soil to determine effects of land disposal on crop growth and uptake of cadmium and radium. In one experiment by Mays, soil was planted sequentially with corn, wheat, and soybeans, over a 1–5 year period (Mays and Mortuedt, 1986). The result suggested that phosphogypsum may be applied to agricultural soils at relatively high disposal rates without increasing levels of cadmium or radioactivity in corn, wheat, or soybean grain. Lindekem considered a hypothetical case of heavy long-term applications of phosphogypsum to soil used for growing food crops (Lindeden, 1980). This was assumed to be an initial gypsum application of 25 tons per hectare followed by alternate year applications of 13 tons per hectare where the radium concentration of the gypsum is 0.6 Bq/g and the till depth is 15 cm. Radium build-up in the soil would reach 0.2 Bq/g after about 100 years. An agricultural worker would receive 150 μSv/y by spending 40 h a week in such a field, while radium uptake by food crops planted in this hypothetical soil were estimated to result in a 50-year integrated dose to the bone surface of 14 mSv, based on the conservative assumption that an adult's total vegetable diet comes from this source.

SUMMARY

The mining, processing, and use of fertilizers containing phosphate in massive quantities has lead to the redistribution throughout the environment of large amounts of uranium, radium and radium decay products. The resulting potential radiological impacts are due to direct exposure, surface run-off, inhalation, and ingestion of foods grown with fertilizers. Gypsum, a by-product of the production of phosphate fertilizers, contains much of the radium originally in the phosphate rock. The resulting waste gypsum piles are a source of radon emission to the environment. There may be additional radiological impact if this waste is used in building materials or as a soil conditioning agent.

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COMPREHENSIVE RADIOLOGICAL SURVEY OF KRESS CREEK WEST CHICAGO AREA, ILLINOIS

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FINAL REPORT

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decreased with increasing distance from the creek. The highest level of total thorium measured in a sample was 555 pCi/g, with a number of other samples exceeding 200 pCi/g. Many of the highest levels were detected in areas near the storm sewer outfall, and hence constitute a potential source of continuing contamination for locations further downstream. The baseline soil in this area contains an average of 1.6 pCi/g of naturally occuring thorium.

Direct levels of radiation measured at 1, 5, 10, and 25 meters from the edge of the creek and 1 meter above ground surface averaged 28, 25, 21, and 14 μR/h (microroentgens per hour) respectively. However, radiation levels greater than 100 μR/h were detected in several locations. Normal background radiation levels in this area averaged 8.6 μR/h.

The contamination levels found along the creek exceed the environmental standards promulgated by EPA under the authority of the Atomic Energy Act of 1954, as amended, for thorium processing wastes. The NRC is charged with implementation and enforcement of these standards. The contamination levels also exceed the identical standards established for cleanup of vicinity properties under Title I of the Uranium Mill Tailings Radiation Control Act of 1978, as amended. The EPA has stated that these standards are appropriate for cleanup of offsite vicinity properties. Therefore, cleanup of the radioactive contamination in the environment is required.

Accordingly, NRC intends to require Kerr-McGee to submit an action plan for cleanup of the contaminated areas. The plan will be subject to review and approval by the NRC staff. The action plan should establish the methods, scope, and timing of cleanup activities.

Copies of this report will be sent to property owners along Kress Creek.

Copies of this report are also available at no charge to members of the public at the West Chicago Public Library.

ANDREWS OFFICE PRODUCTS CAPITOL HEIGHTS, MD (K)

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October 1982

Final
Environmental Impact Statement
for
Remedial Action Standards
for
Inactive Uranium Processing Sites
(40 CFR 192)

Volume I

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cleanup costs, and health benefits. For B3 and B4, which include a range over which remedial action is optional, the cost estimates were derived by assuming a value within the range which would typically be achieved and costing controls to reach this level. For B3, we assumed that at least 0.015 WL (including background) would be achieved. For B4, we assumed that at least 0.03 WL would be achieved.

The extent of contamination of buildings as well as the cleanup costs will not be known in detail until the cleanup program is well underway. Therefore, we used the Grand Junction remedial action program as the basis for our estimates. Appendix B contains a summary of the Grand Junction experience and the cost calculations which support the estimates in Table 7-1.

The cost estimates for each alternative standard are determined by the number of buildings requiring remedial work and the cost per building. As the remedial action criterion is lowered, more buildings will need to be cleaned up, increasing costs. A lower criterion also increases the cleanup costs per building since this requires more complete tailings removal. In many cases, successive actions are needed when the first remedial action does not meet the cleanup criterion. Using active measures to meet a cleanup criterion when the level is only slightly exceeded is much cheaper than tailings removal, roughly one-tenth as costly.

The benefit of cleaning up contaminated buildings is expresed by the number of lung cancer deaths avoided. This is estimated by assuming the risk factors discussed in Chapter 4 are appropriate, an initial distribution of decay product levels in contaminated buildings identical to that for the buildings monitored in Grand Junction, a 50-year average useful life remaining for the stock of contaminated buildings, and a 3-person household size. Also, benefits of cleanup are expressed by the maximum residual risks to people living in the buildings. This risk to an individual is calculated assuming lifetime exposure to radon decay products at the highest level each alternative standard allows.

7.2 Alternative Cleanup Standards for Near-site Contaminated Land

We have analyzed four alternative cleanup standards for near-site (on the site or adjacent to the site) contaminated lands. All have requirements that limit the amount of radium contamination because the presence of radium is a reasonable index of the health hazard, including that due to toxic chemicals as well as other radionuclides.

Alternative L1 approaches a high-cost nondegradation alternative; below this proposed radium limit it is usually not possible, using conventional survey equipment, to accurately distinguish between contaminated land and land with high naturally-occuring levels of radium. Alternatives L2 and L3 approximate optimized cost-benefit standards, but L2 demands a more rigorous cleanup of the soil

TABLE 7-1. COSTS AND BENEFITS OF ALTERNATIVE CLEANUP STANDARDS FOR BUILDINGS (in 1981 dollars)

Alternative Standards	Radon Decay Product Limit (WL)[a]	Number of Buildings Requiring Cleanup[b]	Total Cost (millions of) dollars)	Deaths Avoided (in first 30y)[c]	Estimated Residual Risk of Lung Cancer[d]
B1	0.015	370	11.5	65	0.8 in 100
B2	0.02	330	8.5	60	1.3 in 100
B3	0.005 (above background) to 0.02	420	9.0	65	1.3 in 100
B4	0.01 (above background) to 0.05 (above background)	350	9.5	55	5 in 100

[a]The specified value includes background unless otherwise noted. Background in Grand Junction is approximately 0.007 WL.

[b]See Section 3.4. For Alternative B4, which is identical to the Grand Junction criteria for action, we assumed the geometric mean of our two extreme estimates for the number of buildings requiring remedial action. Assuming the distribution of radon decay product levels will be the same as in Grand Junction, the number of buildings in the United States requiring action was adjusted for the other options.

[c]Based upon the relative risk model. Estimates based upon the absolute risk model are a factor of two lower. Health benefits attributable to reductions in gamma radiation levels are much smaller and have not been quantified.

[d]Lifetime risk to the individual living in a house at the radon decay product concentration limit. This risk is calculated after subtracting background from the level permitted by the standard.

surface. Standard L4 is a least-cost alternative that allows high radiation levels that are close to Federal Guidance recommendations for exposure of individuals to all sources of radiation excepting natural background and medical uses.

The four alternative standards are:

Standard L1. (The standard proposed in April 1980). Land should be cleaned up to levels not exceeding an average 5 pCi/g of radium-226 in any 5-cm layer within 1 foot of the surface and in any 15-cm layer below 1 foot of the surface.

Standard L2. Land should be cleaned up to levels not exceeding an average of 5 pCi/g in the 15-cm surface layer of soil, and an average of 15 pCi/g over any 15-cm depth for buried contaminated materials.

Standard L3. Land should be cleaned up to levels not exceeding an average of 15 pCi/g in any 15-cm depth of soil.

Standard L4. Land should be cleaned up to levels not exceeding an average of 30 pCi/g in any 15-cm depth of soil.

In Table 7-2 we list the estimates of the costs and benefits of each alternative standard for near-site contamination around inactive tailing piles. In each standard, the only remedial method for which we estimated cost was the removal and disposal of contaminated soil, since this is generally less costly than placing earth cover and vegetation over contaminated areas and excluding access by fencing. The benefits are expressed by (1) the number of acres of land that are cleaned up and returned to productive use, and (2) the typical maximum residual risk to individuals living in houses that might then be built on this land.

The number of acres requiring cleanup under each option was based upon the results of the EPA gamma radiation survey of twenty inactive mill sites (Table 3-4). By assuming a typical depth profile of the radium contamination, it is possible to relate the gamma radiation levels measured by the survey to the areas of land contaminated above a specific concentration level of radium. If the top 15-cm layer of earth is uniformly contaminated with 30 pCi/g of radium, the gamma field at the surface would be 63 percent of the gamma flux from an infinitely thick layer, or 34 microroentgens/hr (He78). However, if the 30-pCi/g average in the top 15 cm of earth is due to a thin surface layer of nearly pure tailings of a few hundred pCi/g, the resulting gamma radiation at the surface would be about 54 microroentgens/hr. Since we expect windblown contamination profiles to be somewhere in between these extremes, we estimate that, on the average, 44 microroentgens/hr above background (385 mrem/y) implies 30 pCi/g radium contamination in the top 15 cm of soil (Standard L4). Similar analyses for Alternative Standards L1, L2, and L3 result in 3. 7 and

TABLE 7-2. COSTS AND BENEFITS OF ALTERNATIVE CLEANUP STANDARDS FOR LAND
(in 1981 dollars)

Alternative	Radium-226 Soil Concentration Limit (pCi/g)	Number of Acres Requiring Cleanup[a]	Total Cost (millions of dollars)	Estimated Residual risk of Lung Cancer[b]
L1	5	2700	21	2 in 100
L2	5 to 15	1900	14	2 in 100
L3	15	900	7	6 in 100
L4	30	250	2	10 in 100

[a] Areas of land near inactive tailings piles that have radium contamination in excess of the soil concentration limit.

[b] The lifetime risk of lung cancer to the individual living in a house built on land contaminated to the limits allowed by the alternative standards. This is based on the relative-risk model; use of the absolute-risk model gives risks which are about a factor of two lower.

22 microroentgens/hr, respectively (or 26, 61, and 193 mrem/y, respectively). Additional deeper contamination would yield only slightly higher gamma values because of shielding by the surface layer.

Using these correlations between radium contamination levels and gamma radiation levels, the areas requiring cleanup under each standard were estimated based on the EPA survey data. The total costs of cleanup were then calculated assuming a cleanup cost of $7650 (1981 dollars) per acre. This cost was estimated from EPA field experience (a cleanup program at the Shiprock mill site) and is in agreement with cost estimates of DOE contractors. Areas of heaviest contamination, such as the ore storage area and mill buildings, are excluded from this analysis since we have included them in the analysis of disposal costs for the piles.

The highest risk to people living in houses built upon contaminated land is due to the inhalation of radon decay products from radon that seeps into the house. In the worst case, Standards L1 and L2 would allow thick-surface earth layers with 5 pCi/g contamination, while Standards L3 and L4 would allow thick layers of contaminated soil at 15 pCi/g and 30 pCi/g, respectively. On the average, houses built on such 5 pCi/g earth would be expected to have indoor radon decay product levels of about 0.02 WL. Houses with poorer-than-average ventilation would have higher levels, while well-ventilated houses would have lower levels. Houses built on land more heavily contaminated than 5 pCi/g would have higher average indoor decay product levels in proportion to the contamination. The estimated risks due to lifetime exposure from these levels are listed in Table 7-2. These are maximum estimates since most contaminated land away from the immediate mill sites (where houses might be built) has only thin layers (a few tens of centimeters) of contaminated material.

The gamma radiation levels to individuals permitted under the four alternative standards are 80 mrem/yr for L1 and L2, 240 mrem/yr for L3, and 470 mrem/yr for L4. This assumes a thick layer of contaminated material over a large area at the maximum permitted levels of radium concentrations. These doses would lead to increased risk of many kinds of cancer, but this increase would be small compared to the lung cancer risks due to radon decay products.

7.3 Alternative Cleanup Standards for Offsite Properties

Tailings on offsite properties which are not associated with building construction are usually there because someone transported them from a tailings pile. Examples of this kind of misuse are tailings used as fill around fence posts and sewer lines, as the basis for sidewalks and driveways, and as conditioners for soil in gardens. Most tailings misused in this way are still concentrated; they are not diluted by large quantities of earth or spread thinly over large areas.

The major hazard stems from the chance that indoor radon levels will be high in new buildings constructed on contaminated offsite properties. There could also be a significant gamma radiation hazard if people spend a lot of time close to the tailings.

We expect that offsite properties where tailings were misused will typically exceed all the radium concentration limits specified for land contamination in Alternative Standards L1 through L4. Therefore, virtually all of the 6500 contaminated sites identified in Chapter 3 would require cleanup under any standard. Based on engineering assessments and similar cleanup work near a mill site in Edgemont, South Dakota, we estimate it would cost $6,000 to clean up each of these properties. This implies a total cleanup cost of $39 million. However, many of these sites are unlikely to cause a significant present or future hazard, either because of their location or because the quantity of tailings involved is so small. Cleaning up such sites implies high cost without significant benefits.

It is consistent and simple to use the same numerical cleanup criteria for offsite contamination of properties as for near-site land contamination. Since some offsite contaminated properties present a minimal hazard and would cost a great deal to clean up to any reasonable radium concentration criterion, additional criteria are considered in one of the following alternative standards for contaminated offsite properties:

Standard P1: Offsite properties should be cleaned up to the same levels as near-site land,[1] with no exceptions.

Standard P2: Offsite properties should be cleaned up to the same levels as near-site land, with the following exceptions:

a. When contamination levels averaged over 100 m^2 are less than the action levels required for near-site lands.
b. When the hazard from the tailings is judged to be insignificant because of location.

Small amounts of tailings will be eliminated from consideration if levels are averaged over an appropriate area. For Standard P2 we have selected 100 m^2 as a reasonable area for this purpose since this is the typical area of the foundation of a house. Thus, risk levels allowed under Standard P2 should be no higher than the risks allowed under the corresponding near-site land cleanup standard. Additional sites will be eliminated under Standard P2 because of their location.

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[1] Alternative Standards L1, L2, L3, or L4; whichever is selected as a land cleanup standard.

From Figure 5-1 we can determine how much radon emission would be reduced by adding one meter of earth. If the only benefit of thicker covers were to reduce radon emissions, we would find the cost-effectiveness of each additional meter of earth to be considerably less than that of the first meter. But thick covers have additional benefits: they last longer than thinner covers and are barriers against intrusion. Therefore, the net benefits of reducing radon emissions cannot be isolated.

The disposal cost analysis in Chapter 6 applies only under the stated assumptions. If local earth near a pile is very sandy, or if suitable earthen materials are not available nearby, then satisfying the Proposed Standard and Alternative A, which have the strictest radon emission control level, could require several additional meters of cover. Conversely, if earthen materials are more easily available or of higher quality (i.e., clays) than is assumed, the costs will be lower. Because of the lack of full-scale disposal experience, however, there is a greater risk of the cover requirements for the Proposed Standard and Alternative A being significantly underestimated than for Alternatives B through E.

NRC (NRC80) has evaluated the potential environmental impacts of obtaining cover materials in regions where uranium is mined. As a rule, the environmental impacts will be greatest for the Proposed Standard and Alternative A, less for Alternative B, and least for Alternatives C through E. Even under relatively unfavorable conditions, however, the effects are largely temporary; the longest-lasting effects are changes of topography at borrow sites for the cover material. This issue is highly site-specific, however, and definitive information on the environmental effects of obtaining cover materials at the 24 inactive sites is not yet available. We expect such effects will be small overall, but the Proposed Standard and Alternative A are the most likely to cause significant temporary environmental disturbances.

Form of the Radon Standard

We have expressed the radon limit in terms of the release rate per unit area from the tailings. However, a number of alternative criteria could be used to control radon emissions from the piles:

a) dose rate limits for individuals or populations, (mrem/y, person-rem/y, person-WLM/y),
b) radon concentration limits in air (pCi/l),
c) total radon release rate limits (pCi/s), and
d) release rate limit per unit area (pCi/m^2s).

Because short-term fluctuations are unimportant, we will consider all of these as annual averages. Radon emissions from tailings to the air cannot be separated from those from a cover or normal land, however. Therefore, a standard using any of these criteria must apply to either the total radon release rate from the surface of a pile or to the radon release rate from tailings with allowance being made for the radon

(1) Radium concentration is directly related to the hazard of most tailings. (Occasionally it is not sufficient where other specific radioactive or toxic elements in uranium ore processing residues have been concentrated.) Quantities (2), (3), and (4) result directly from the radium in tailings.

(2) Gamma radiation levels can be conveniently measured, but they are related to only part of the hazard. Tailings that are covered with a few feet of earth could satisfy a gamma radiation standard, yet be hazardous to build upon because of radon emissions.

(3) Radon emission is usually the principal hazard from uranium mill tailings. Radon release rates vary greatly with changes in weather and soil moisture, however. A standard based on the radon release rate would require repeated measurements over varied conditions to determine meaningful averages.

(4) The predicted radon decay product concentration is related to the hazard, but estimates of the indoor radon decay product concentrations are very uncertain. Furthermore, either the radium concentration or radon release rate from the land must first be determined to make such estimates, so (4) offers no advantage over (1) or (3).

8.3.2 Preferred Cleanup Standard for Land

We prefer Alternatives L2 and P2 as cleanup standards for near and distant land, respectively. Specifically, land should be cleaned up to levels not to exceed an average of 5 pCi/g of radium-226 in the first 15 cm surface layer of soil and an average of 15 pCi/g of radium-226 in any layer of 15 cm depth at deeper levels. Offsite properties should be cleaned up to these same action levels, with the following exceptions:

a) when contamination levels averaged over 100 m^2 are less than these action levels; or

b) when the hazard from the tailings is judged to be not significant because of their location.

A 5 pCi/g limit over the first 15 cm can be easily implemented with relatively low cost gamma radiation survey methods. For tailings below 15 cm, the concentration limit of 15 pCi/g is also easy to implement. Alternative L1 would require more skill and training of personnel, and greater use of expensive measuring techniques, but cleanup would only be marginally more complete. Very thick deposits of

material with up to 15 pCi/g of radium-226 generally would be hazardous o build on, but are unlikely to occur. A concentration of 15 pCi/g is ikely to occur only in thin layers at the edges of more concentrated deposits that would be cleaned up under a 15 pCi/g criterion. Under most foreseeable circumstances, we believe the residual hazard would be acceptably low under Alternative L2.

Alternatives L3 and L4 do not take full advantage of practicable cleanup. Several thousand acres next to disposal sites would require land-use controls. The costs saved are small in relation to total costs and do not warrant the higher risks that would remain.

We believe it is neither practical nor worthwhile to cleanup contaminated areas to surface concentrations below 5 pCi/g. Identifying contaminated surface soils with radium concentrations less than 5 pCi/g is difficult and expensive. Complex measurement techniques are required. Increasingly large land areas would need to be cleaned up. Doing this would provide very little gain in health protection, because such slightly contaminated soils are usually thin layers containing small amounts of tailings that pose insignificant risks.

For offsite properties, the cleanup costs vary little with the choice of numerical cleanup standards because tailings typically have been used with little mixing with other materials. If a standard based on Alternative L2 for nearby land is rigidly applied, up to $39 million may be spent in cleaning up these properties. However, many of these contaminated offsite properties present little existing or potential hazard because of the small amount of tailings involved, or because of their location. In Chapter 7 we considered applying the land cleanup standard for offsite locations only when appropriate threshold conditions are exceeded. This was projected to save $24 million without sacrificing protection of people. We therefore selected this alternative.

Radiation Hazards not Associated with Radium-226

Radium-226 concentrations in the residual tailings may not adequately measure the radiation hazard in all cases. The possibility that this could happen at one or more inactive processing sites cannot be ruled out, but we do not know of a site where this has happened. Should such circumstances occur, our supplemental standards (see below) will require the implementing agencies to reduce residual radioactivity to levels that are as low as may reasonably be achieved.

8.4 Supplemental Standards

In view of the varied conditions and our limited remedial action experience with tailings, these standards must be flexible. We believe our standards are the most protective that can justified for general

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CLEANUP STANDARDS FOR RADIUM CONTAMINATED SOILS

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Abstract

In 1983 EPA promulgated standards for cleanup of uranium and thorium mill tailings at 40 CFR 192. These standards address a specific example of the cleanup of radium contamination. They have been used for the cleanup of radium-contaminated soils at other sites, primarily because they are the only related standards that exist. However, EPA advised caution at the time these standards were issued: "It should be noted that these standards in no way are intended to establish precedents for other situations or regulations involving similar environmental objectives, but with different economic and/or technological circumstances."(EPA 83) This paper assesses the suitability of these standards for use in the cleanup of contaminated soil at sites other than uranium or thorium mill tailings sites.

The 40 CFR 192 Cleanup Standards

The 40 CFR 192 rules specify two types of standards. The first addresses the disposal of uranium and thorium mill tailings, and are not discussed in this paper. The second addresses cleanup, and are the subject of this paper. They include limits for indoor radon concentrations and indoor gamma exposure rates for cleanup of buildings, as well as limits on radium concentrations in soil for cleanup of land. The former, those for indoor radon and gamma exposure, are health-based standards; while the latter, radium concentrations in soil, are technology-based standards, keyed to the sensitivity of radiation monitoring systems. For uranium tailings, the increased indoor radon concentrations and indoor gamma exposure rates were caused by placing tailings around buildings and houses. Radon is a decay product of radium and, since it is an inert gas, can move through soil and enter buildings above soil that is contaminated with radium. It was assumed at the time the 40 CFR 192 standards were promulgated that the indoor standards would be achieved by removing such tailings and replacing them with clean soils.

The 40 CFR 192 standards for soil specify a concentration limit of 5 pCi/g radium in the top 15 cm of soil and 15 pCi/g radium in any 15 cm thickness below the top 15 cm. The limit for

the top layer was based on limiting external exposure rates to persons who may spend time on the land. Its purpose is to indicate when cleanup of thin surface layers of windblown tailings is necessary to provide adequate public health protection. Although this criterion provides adequate health protection for the situations it was developed to address, the value was selected with the limitations of field measurement equipment in mind, and the transient nature of windblown contamination situations.

The 15 pCi/g soil concentration limit is a technology-based standard. It is a practical measurement criterion for use in locating discrete quantities of tailings that were deposited or placed in subsurface locations at mill sites. These tailings deposits are generally limited in area and volume, with little or no mixing with adjacent soils, and have activities exceeding 100 pCi/g. Convenient measurement techniques for assaying radium activity in boreholes can not readily achieve a sensitivity better than 15 pCi/g in 15 cm layers. Since this is adequate for locating the edge of subsurface deposits of uranium mill tailings, it was selected as an appropriate standard for use at the tailings sites. Cleaning up deposits of tailings using this standard will leave at most only very small deposits that would not produce sufficient radon to cause a significant increase in indoor levels in a structure built over them.

The Relationship Between Indoor Radon and Radium in Soil

In this paper it is assumed that the goal of land cleanup around houses should be to meet the health protection standards of 40 CFR Part 192. These require limiting the average indoor radon concentration to 0.02 WL (4 pCi/l) including background, and restricting the indoor gamma exposure rate to 20 microR per hour above background at any location in a permanently occupiable structure. The technical objective, therefore, is to achieve those conditions in the soil around present (and potential) occupiable structures that will satisfy these indoor requirements.

The characteristics of the soil, the pressure differential between indoor air and the atmosphere, and the air exchange rate of the building itself are major factors that determine the buildup of indoor radon. For this paper, a model called RAETRAN (Ro 89) was used to examine the relationship between radium concentrations in soil and indoor radon concentrations in a house constructed over land contaminated with radium. In the RAETRAN estimates, the soil characteristics were varied, as were the radium concentrations in soil. The pressure differential and air exchange rates were held constant at values representing a new house constructed to meet current energy conservation guidelines.

RAETRAN predicts the movement of radon in soil by both diffusion and advection. Radon moves through soils along the path

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Remedial Planning Field Investigation Team (REM/FIT)

ZONE II

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CH2M HILL
Ecology &
Environment

REMEDIAL INVESTIGATION
REPORT

Kerr-McGee Radiation Sites
West Chicago, Illinois

WA No. 82-5L94.0

September 29, 1986

- Residuals containing 92 pCi/g of Th-232 covering an area of about 20 square feet to a depth of 4 inches (10 cm) produce an exposure rate of about 20 microR/hour at 3 feet (1 m) above the surface. The gross reading in this situation would be about 30 microR/hour (e.g., 20 + 10 background).

- The gamma exposure rate for a 4 inch thick plane source of infinite lateral extent, with a concentration of 15 pCi/g is about 30 microR/hour, or a gross (including background) measurement of 40 microR/hour.

- The exposure rate from a deposit of thorium residuals with a Th-232 concentration of 60 pCi/g covering an area of 50 square feet to a depth of 4 inches would be about 30 microR/hour.

Based on these observations and the assessment in Appendix F, it is concluded that the soil sample results reported by Kerr-McGee are not specifically incompatible with the report by Kerr-McGee (Denny, 1986) and that mitigations were performed according to the specified exposure rate cleanup criteria (maximum of 30 microR/hour, gross measurement). The soil sample results reported by Kerr-McGee (e.g., up to 92 pCi/g of Th-232) are not incompatible with the hypothesis that mitigations met the criteria of 15 pCi/g of Th-232, averaged over a 6 inches (15 cm) depth and 1000 square feet (100 m^2) area (EPA, 1983). However, the sample results indicate residuals were left at some locations, and application of mitigation to as-low-as-reasonably-achievable (ALARA) may not have always been performed. Furthermore, it is noted that the soil sampling results would have been more useful if they had been collected in accord with the EPA depth and area criteria.

of early August about 7,000 cubic yards of contaminated material remained at the STP.

Residuals from the REF have also been identified at 117 additional sites throughout the West Chicago area. These sites lie primarily east of the REF. Kerr-McGee had removed known residuals from nearly all of these sites within the City of West Chicago by 1985. Surveys of the contaminated sites outside the City limits have been made, but these contaminated sites have not been remediated.

DATA ASSESSMENT AND CONCLUSIONS

There are several routes of potential risks to the environment and public health; including direct external radiation exposure; inhalation exposure; and ingestion of contaminated soils, groundwater, and surface water. The contaminated media at the subject sites are wastes from the REF mixed with soils on the subject sites.

The hazardous characteristics of the thorium residuals are primarily due to the radioactive constituents. The potential for release of heavy metals to the groundwater appears to be minimal, based on the RI activities and assessments. Specifically, validation tests using the EP Toxicity Test to determine the leachability of hazardous substances indicate a low potential for significant groundwater pollution.

The primary radionuclides present are thorium-232 and uranium-238 and their associated decay products. The principal potential risks to man include external gamma radiation exposure, and radiation exposure from inhalation of airborne decay products of thoron (Rn-220) and radon (Rn-222). The REF wastes, which are the original source of the contamination, contain nominal concentrations of Th-232 up to 4000

pCi/g with U-238 concentrations of about one-tenth the Th-232 values. The decay products of Th-232 and U-238 in the wastes are generally in radioactive equilibrium.

HEALTH RISKS

Peak external gamma exposure rates in areas accessible to the public at STP range up to over 1000 microR/hour. This area is isolated, but open to limited access. Most areas at RKP outside of the fenced area, where there is common public access, are around 30 microR/hour or less. The average concentrations of Th-232, U-238, and the associated decay products are about an order of magnitude less than the indicated values for the original source material (REF wastes). Within the fenced area, which is not guarded, peak exposure rates exceed 1000 microR/hour.

The public health impacts imposed by the thorium residuals at RKP, STP, and off-site Properties are principally related to their potential and actual radiation exposure of people on and near the sites. The potential health risks resulting from these impacts are the possibility of human cancers and genetic anomalies associated with ionizing radiation. The present impacts upon groundwater appear to be negligible.

The risks for cancer incidence for this assessment are given as excess fatal cancers. The risk for total cancer incidence (fatal plus nonfatal) would be somewhat larger. The risks are excess risks due to the designated situation. Thus, the risk for an individual is the sum of the normal incidence of cancer including the risk from exposure to natural background radiation, the additional risk due to radiation exposures from the thorium residuals, and the risks to any other special causes of cancer the person is exposed to. All of these causes of cancer have statistical variability or uncertainty.

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Final
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for
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(40 CFR 192)

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Although the Church Rock tailings-dam failure occurred spontaneously, natural events could also precipitate such a failure: most notably severe flooding or an earthquake. In Chapter 8, the probabilities of such events are discussed, along with engineering and site selection options for minimizing these probabilities. Also discussed in Chapter 8 are the impacts of events such as tornadoes and glaciation on the effectivness of contaminant controls.

3.3.2 Misuse of Tailings Sands

In the recent past, uranium mill tailings have been used extensively as a building material, chiefly as fill around and under foundations and concrete slabs. The tailings sands have ideal physical characteristics for this purpose. However, such use typically results in building occupants being exposed to high levels of radon decay products and thereby incurring a significant lifetime risk of lung cancer. In Grand Junction, Colorado, over 700 buildings have been identified as contaminated and requiring remedial action. In other mill towns, it is estimated that more than 350 buildings are contaminated. In addition to buildings, many thousands of other locations have been identified (e.g., sidewalks, lawns, gardens, driveways) in mill towns where tailings have been used. These buildings and locations were contaminated by tailings from inactive mills. We have not assessed the extent of existing misuse near active mills.

3.4 Environmental Releases from Heap-Leaching Operations

The principal solid waste from heap leaching is the barren material remaining after uranium recovery. Airborne emissions from heap-leaching operations include particulates suspended by wind erosion of the pile and radon gas. The particulates will contain toxic elements and radionuclides in proportion to the ore concentrations. The amount of radon and particulates given off will be proportional to the size of the operation. These have been calculated for the heap-leaching cell covering about 0.5 acre in area described in Chapter 2.

Particulate emissions from the dry portion of a heap-leaching cell are estimated to be about 1 MT annually. The radon emanation rate from this operation is calculated to be 25 Ci/y (NRC78). This is less than one-half as much as a tailings pile per unit acre.

Releases of contaminants to groundwater could result from the seepage of leachate containing elevated concentrations of radionuclides and toxic elements. This, however, would not normally pose a problem during operations since an efficient heap-leaching operation requires an impermeable pad and all leachate is collected for processing. After termination of operations, normal rainfall could lead to some leaching from the piles, but we expect this to be no greater threat than leaching from an unstabilized conventional tailings pile.

A general expression for the thoron (or radon) source term (Ro81) is

$$Q = R \rho_b E \lambda$$

where

R = radium-224 content (pCi/g)
ρ_b = bulk density (g/m^3)
E = emanating power
λ = decay constant for thoron

Since the same equation holds for radon ($\lambda = 2.1 \times 10^{-6}$/sec), the thoron ($\lambda = 1.25 \times 10^{-2}$/sec) production rate will be in the same ratio as the decay constants or nearly 6,000 times larger, for the same radium density, than that for radon. The higher thoron production rate is, however, offset by its shorter half-life since, once the thoron has decayed, it no longer migrates freely through the tailings or cover material. For example, the thoron flux at the surface of a bare tailings pile containing 280 pCi/g of radium-224 is about 21,600 pCi/m^2s rather than the 280 pCi/m^2s of radon for the same radium-226 density. The analytical techniques described in Section 8.3.1 may also be used to determine the thickness of an earthen cover required to attenuate the thoron flux. Based on the nominal thoron flux value given above, some typical thoron reduction thicknesses are shown in Table G-1.

Table G-1. Estimated Cover Thickness (in meters) to Reduce Thoron Emissions to 20 pCi/m^2s

Thoron Emission from Tailings (pCi/m^2s)	Percent Moisture Content of Cover 6	8	10	12
10,000	0.0826	0.0637	0.0491	0.0379
20,000	.0918	.0708	.0546	.0421
30,000	.0972	.0750	.0578	.0446
40,000	.1010	.0779	.0601	.0463

The epidemiology and dosimetry of thoron and its decay products have recently been reviewed by the International Commission on Radiological Protection (ICRP81). They concluded that, because of the short half-life of thoron and its first decay product, the radiation hazard from the short lived progeny of thoron is normally well represented by the potential alpha energy exposure of lead-212 and bismuth-212. Based on this conclusion, the effective dose equivalent for the thoron decay products is about one-third that of the short-lived radon decay products.

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Thoron in the Environment

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Thoron (^{220}Rn), an isotope of the radon family, is produced in the earth's crust at a rate comparable to that of common radon (^{222}Rn). Thoron's average activity concentration in soil gas and ground-level outside air is comparable to that of radon. Recent data from Europe and the United States indicate that in terms of the energy of the alpha particle decays of thoron's progeny, its concentration in indoor air is significant, typically about half that due to radon progeny. This paper reviews current knowledge about thoron and its progeny in the outdoor and indoor environments and discusses issues involved in assessing whether or not it is a significant indoor pollutant.

In environmental science the term radon is usually taken to refer to the specific isotope ^{222}Rn of the radon isotope family. Airborne ^{222}Rn has been receiving a lot of attention as a pollutant, particularly at the elevated levels that can be found in the indoor air of some housing. Another member of the radon isotope family, ^{220}Rn, commonly called thoron, is produced in the earth's crust at rates comparable to that of radon. Its chemical properties are the same as that of ^{222}Rn, and it also can migrate to the earth's atmosphere where it is found in abundance near the ground. Thoron is an alpha particle emitter as is radon. Why have we not heard more about it as an important agent in the atmospheric environment and as a possible pollutant? Should we hear more? To address these questions, we first review some basic scientific information about this important radionuclide produced in the earth's crust.

Diffusion of Thoron to the Atmosphere

In collaboration with Ernest Rutherford, thoron was discovered in 1899 by R. B. Owens at McGill University. At this early stage in the study of radioactivity (the existence of the nucleus in the atom had not yet been established) the gas was described as *thorium emanation* since about all that was clear was that it was radioactive gas emitted from thorium-bearing minerals. In fact, thoron does not come directly from thorium but from the radium isotope ^{224}Ra, a decay product in the thorium (^{232}Th) series. Thoron's half-life is 56 s, and upon decay it emits a single alpha particle of energy 6.3 MeV. In comparison, ^{222}Rn has a half-life of 3.3×10^5 s (3.8 days) and emits a 5.5 MeV energy alpha particle. There are additional radioactive isotopes after thoron in the thorium decay series which terminates in the stable isotope ^{208}Pb (see Figure 1).

The origin of the thoron in the atmosphere is almost entirely from diffusion from the top few centimeters of soil where thoron is produced as a decay product from the trace amounts of ^{232}Th pervasive in almost all soils and rocks. The combination of small thorium content of sea water, and short migration distance of thoron in water, make the contribution from the oceans negligible. It has sometimes been proposed that direct release from plants, for example by transpiration, is an important mechanism for release of thoron to the atmosphere.[1] Dr. Stewart Whittlestone of the Australian Nuclear Science and Technology Organization and I recently tested this assumption in an extensive survey of flux from Australian soils[2] (see Figure 2). We found no evidence of a significant effect. In fact, increased vegetation tended to correlate with slightly reduced release of thoron to the atmosphere, perhaps due to a modest blocking effect of the surface layer of moist, organic materials of low radioactivity content.

Although deeper layers of soil also produce thoron, the thoron is unable to diffuse to the surface before experiencing decay to non-volatile decay products which readily attach to soil particles. Not all thoron that is produced in the top few centimeters of soil escapes to the atmosphere. In order to diffuse to the atmosphere, thoron must first be released to the pore space of the soil. It is only when the parent, ^{224}Ra, is close enough to the surface of a grain that thoron atoms can recoil into the air of the pore space where they are free to diffuse: thoron atoms whose recoil paths terminate in a grain almost always will be trapped in place and unable to migrate before decay. The fraction of atoms reaching the pore space is called the emanating fraction, and surprisingly, it is typically about a third.[2] This is much more than would be expected if the parent radium were uniformly distributed throughout the interior of spherical grains since a typical recoil range for thoron atoms in minerals is of the order of 30 nanometers.[3] Much of the trace amounts of thorium in soil and rock originates from precipitation from matrix water; thus there is typically a preferential deposit of thorium minerals on the surface of soil grains. Surface irregularities of nonspherical grains and moisture interaction with grain surfaces provide further factors enhancing release of thoron to the pore space.

In terms of the production rate of radon and thoron, the average trace concentrations of ^{226}Ra and ^{224}Ra in the soil are comparable (producing on the order of 25 atoms per second per kilogram). The mechanisms for radon release to

Implications

Present evidence indicates the equivalent radiation dose from thoron and its progeny is about 10 to 20 percent of that due to radon and its progeny in the indoor environment. Assuming current estimates for the health effects of radon are valid, this suggests thoron is potentially a significant indoor pollutant, although clearly less important than radon. Uncertainties are still large in the dose estimates for thoron progeny, the distribution of thoron among U.S. housing, and the mode of entry of thoron into indoor airspace. More research is needed to improve our evaluation of thoron as an indoor pollutant and to establish corrective measures should thoron be a significant pollutant.

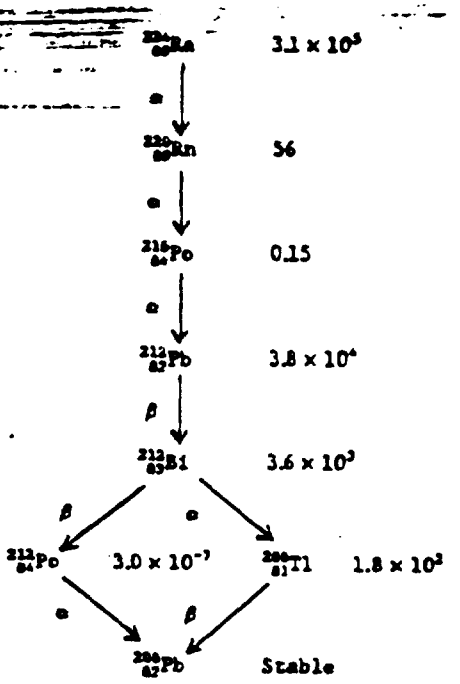

Figure 1. The part of the thorium ($^{232}_{90}Th$) decay series immediately preceding and following thoron ($^{220}_{86}Rn$). Also shown are rounded half-lives in seconds, alpha particles decays (α), and beta particle decays (β).

the pore space are similar to those for thoron, so given radon's longer half-life, radon is able to diffuse from greater depths of soil and produce a larger atomic flux. The result is an average flux density of radon atoms to the atmosphere, about 1 atom cm^{-2} s^{-1}, a factor of almost a hundred greater than that for thoron, about 0.014 atoms cm^{-2} s^{-1}.[4] (In terms of the customary units of activity, the radon flux density is less, about 0.021 Bq m^{-2} s^{-1} versus about 1.7 Bq m^{-2} s^{-1}, but these numbers can be misleading in the present context in the sense that it is the number of atoms that is more directly proportional to the ultimate energy released by decay of the progeny in the atmosphere.)

The diffusive flux of both radon and thoron from soil to the atmosphere can be quantified for homogeneous thick soil with:[5]

$$J_a = (D_e/\lambda)^{1/2}\, \alpha_{Ra}\, \rho\, f \qquad (1)$$

where: J_a is the atomic flux density;
D_e is the effective diffusion coefficient;
λ is the radon or thoron decay constant;
α_{Ra} is the activity concentration per unit mass of the parent radium (Becquerel per unit mass);
ρ is the soil bulk density; and
f is the emanating fraction (dimensionless).

Variation of the density and diffusion coefficient among (dry) soil types is generally modest. Furthermore, the flux density depends only on the square root of the diffusion coefficient. Hence, it is plausible that variation in the emanating fraction and radium concentration is particularly important for causing variation in flux density from one site to the next, and there is some evidence that this is indeed true for both radon and thoron.[2] This issue can be of importance in anticipating what soil locations might be strong sources of thoron flux to the atmosphere. Thus, identification of soil sites possessing high concentration of ^{232}Th, such as by analysis of airborne radiometric data, may not be sufficient to establish high release of thoron. Geological characterization of soil should be useful for predicting thoron release but so far there has been little systematic study of this approach. One exception is that areas of monazite sands are known high sources of thoron.

The fact that the average atomic flux density for thoron is much less than for radon might seem to provide the answer to our original question of why less attention is given in the environmental science to thoron: its release to the atmosphere is much smaller. However, the situation is subtle and this answer is misleading. For the indoor environment of houses, where pollution from radon may be a concern, mechanisms different from the pure diffusion can be important for entry of radon. Even in the outdoor environment, at ground level thoron is proportionally more important than would be expected from flux considerations alone. With its longer half-life, radon is mixed in a much deeper layer of the atmosphere resulting in significant dilution. In terms of decay rate per unit volume of air at the breathing level above ground, the concentration of thoron is typically greater than that of radon. For example, in the continental United States, average activity concentrations for thoron might be of the order of 15 Bq m^{-3} versus 5 Bq m^{-3} for radon (1 Bq m^{-3} = 0.027 pCi/liter), although atmospheric conditions can cause wide variation among these numbers.[4]

Thoron in Indoor Air

Direct measurement of thoron gas is difficult and the number of indoor measurements is quite limited, particularly in comparison with the data available for indoor radon. About the best that can be said at this time is that average concentrations on the ground floor in temperate climates of the United States and Europe are likely to fall in the range of 5 to 30 Bq m^{-3}.[5] Traditionally, it has been assumed that the only mechanism by which significant thoron can enter houses is by release from the internal surfaces of building materials such as concrete or by direct input of outside air through large openings such as windows.[6] It is now well established that elevated levels of radon in houses are primarily due to flow transport from soil through cracks and joints in places such as the basement,[7] driven by pressure gradients caused by convective currents from heating inside a house and the influence of outdoor wind. Even in light of experience with radon, pressure driven flow transport was not considered plausible for thoron since its half-life was felt to be too short to permit time for entry from soil.

For a room with well-mixed air it is easy to quantify the expected steady-state indoor thoron concentration C_{Tn} using:

$$C_{Tn} = \frac{EV^{-1} + u\, C_{Tn,0}}{\lambda + u} \qquad (2)$$

where u is the ventilation rate, $C_{Tn,0}$ is the outdoor thoron concentration. E is the rate at which thoron is supplied to the room, and V is the volume of the room.[5,8] If the mechanism supplying thoron is diffusive flux from inside surfaces, then:

$$E = JA \qquad (3)$$

where J is the diffusive activity flux density ($J = \lambda J_a$) from the inside surfaces with area A. Surprisingly, calculations with Equations 3 and 2 had not been carried out extensively enough to verify the plausibility of the general importance of diffusive flux from inside surfaces. One difficulty is the limited availability of data for J, a quantity which can vary greatly depending on the building material. The thoron flux can also be reduced by any coverings present such as paint, sealant, or tapestries, since, due to thoron's short half-life, even a thin layer of material can be important.

In the mid-1950s work by our research group at New Mexico Tech indicated that flow transport from soil might also be an important mechanism for entry of thoron.[4,9] This conclusion about the possible importance of soil resulted from careful analysis of data on indoor thoron and radon

that we had obtained across the United States over the previous five years. The evidence was indirect, but highly suggestive. Houses with higher radon concentrations tended to have higher concentrations of thoron and its progeny. If thoron were coming from a different source, what was the explanation for this correlation given it had been established that transport from soil was usually the most important factor causing high indoor radon? Secondly, concentrations of thoron and its progeny tended to be lower in upper stories of buildings that were more isolated from the soil. Thirdly, for a house of typical regional construction on the campus of New Mexico Tech subject to intense study, we were able to show that the primary mode of entry of thoron was through cracks in the foundation.[10] Finally, theoretical analysis of flow through representative small cracks and joints in foundations indicated that flow times were often short enough to enable thoron to pass without significant decay.

Figure 2. Dr. Stewart Whittlestone, of the Australian Nuclear Science and Technology Organization, making measurements of the release of thoron from soil in New South Wales.

The exact extent to which flow transport of thoron from soil is important over the range of housing and buildings is not yet established. Calculations using Equation 3 indicate that diffusion from building materials is a feasible mechanism to explain observed indoor levels for certain materials such as unpainted concrete. However, the basic proposition of the likely importance of flow transport from soil through a building's foundation is much less controversial today than it was just a few years ago. Should direct entry from soil turn out to be the major source of indoor thoron there are a number of practical implications. Much of what has been learned from experience with radon would apply to thoron. For one thing, if mitigation is necessary, efforts should be directed at entry from soil not treatment of indoor surfaces. Identification of locales with higher indoor thoron would focus more on soil properties and entry paths rather than materials used in construction. There would probably be a greater tendency for locales with high indoor radon to correlate with locales of high indoor thoron since some important factors such as permeability of soil and pressure gradients in houses would be common for both.

Further discussion of thoron as a possible indoor pollutant requires a brief digression into the physical properties of airborne radon isotopes and their progeny. An important hazard associated with airborne radon isotopes comes not from the inert gases themselves but from their radioactive decay products. Even though non-volatile metals, a large fraction of these isotopes remain suspended in the air (often attached to aerosol) where upon inhalation some of them are deposited in the lung. The primary health hazard is lung cancer due to the densely ionizing alpha particle radiation of these decay products that deposit on the bronchial epithelium of the lung. From the dosimetry standpoint, the important factor in the air that is breathed is the quantity of potential alpha particle energy concentration from these decay products. This quantity is given the acronym PAEC and has dimensions of energy per unit volume of air. Due to historical reasons associated with the mining industry, in the United States the unwieldy unit of Working Level (WL) is used for PAEC, although the proper international unit is Joules per cubic meter ($1 WL = 2.08 \times 10^{-5} J\ m^{-3}$).

If thoron and its progeny were in complete equilibrium, a thoron concentration of 276 Bq m^{-3} (7.5 pCi $liter^{-1}$) would correspond to one WL. However, among the important thoron and radon progeny, the thoron progeny $^{212}_{82}Pb$ has the longest half-life (3.8×10^4 s = 10.6 h) and there is a greater opportunity for it to be lost from an air mass by deposition on nearby surfaces. Hence, disequilibrium between thoron and its progeny is typically much greater than that between radon and its progeny. The extent of disequilibrium can be quantified by the thoron equilibrium factor, approximately given by:[6]

$$F = \frac{0.91\ C_{Pb} + 0.09\ C_{Bi}}{C_{Tn}} \qquad (4)$$

where C_{Pb}, C_{Bi}, and C_{Tn} are the activity concentrations for ^{212}Pb, ^{212}Bi, and ^{220}Rn. A value of 1(0) corresponds with complete equilibrium (disequilibrium). Progeny can further be classified as belonging to one of two broad categories: unattached to aerosol (free atoms and clusters) and attached to aerosol (particles larger than about 10 nm). Given the concentration of thoron inside a room, the physical processes controlling the disequilibrium of progeny and their physicochemical form are the same as with radon progeny, so it is possible to make estimates with models[5,8] based on physical parameters obtained primarily from radon measurements. There has been some verification with thoron and progeny measurements,[10] so these models should at least be valid for indicating general trends. Figure 3 shows a representative calculation of disequilibrium for a constant rate of input of thoron and a ventilation rate of 0.75 h^{-1}. Compared with similar radon progeny calculations, distinctive features include a high degree of disequilibrium among isotopes (a small equilibrium factor in the range 0.002 to 0.06 versus about 0.02 to 0.5 for radon), and, due to its very short half-life, the presence of ^{216}Po primarily in an unattached state. Features similar to those observed with radon progeny are that the other thoron progeny are primarily in the attached state and that the equilibrium factor increases significantly with increased aerosol concentration. Model calculations also indicate that higher ventilation will reduce PAEC for both radon progeny and thoron progeny, but that due to its short half-life thoron itself will typically not vary with ventilation for common ranges of ventilation (if $u \ll 0.693/T_{1/2} = 45\ h^{-1}$). Due to its short half-life thoron may not be distributed uniformly throughout a room, although the longer lived progeny should normally be about as well-mixed as are radon and its progeny.

For thoron progeny, ^{212}Pb is the most important isotope controlling the PAEC. This isotope is relatively easy to measure, for example by conventional filter techniques, so there

are comparatively more data available for PAEC due to thoron progeny than for thoron itself. Although the statistical base is still small, it now seems clear that from the standpoint of PAEC, thoron progeny are a significant contributor to the indoor environment for housing in the United States and Europe.[8,11] Table I summarizes some of the data available comparing the PAEC due to thoron progeny to that due to radon progeny. An average value for the ratio of PAEC due to thoron to that due to radon might be about 50 percent. Furthermore, this ratio remains significant over a fairly wide range of conditions. Modeling suggests that one reason this ratio does not fluctuate more widely is that some of the important removal processes causing variation in PAEC (such as ventilation and deposition on walls) commonly affect both radon and thoron progeny.

Is Thoron a Significant Indoor Pollutant?

If the ratio of PAEC due to thoron to that due to radon is as high and pervasive as suggested by Table I, should there not typically be a health effect due to thoron significant in relation to that identified for radon (whether small or large)? One response relates to the fact that public health concerns with radon have so far tended to focus on houses with elevated, not average, levels of radon. Although such houses will typically also have increased levels of thoron, the amount of increase for thoron is usually not as great as that for radon. In this class of housing the PAEC ratio will thus be typically less than 50 percent. For example, in one recent study[12] data for PAEC due to thoron (PAECTn) and PAEC due to radon (PAECRn) were fit well by a phenomenological relation which, in rounded numbers, is given by:

$$\text{PAECTn} \propto (\text{PAECRn})^{0.4}. \qquad (5)$$

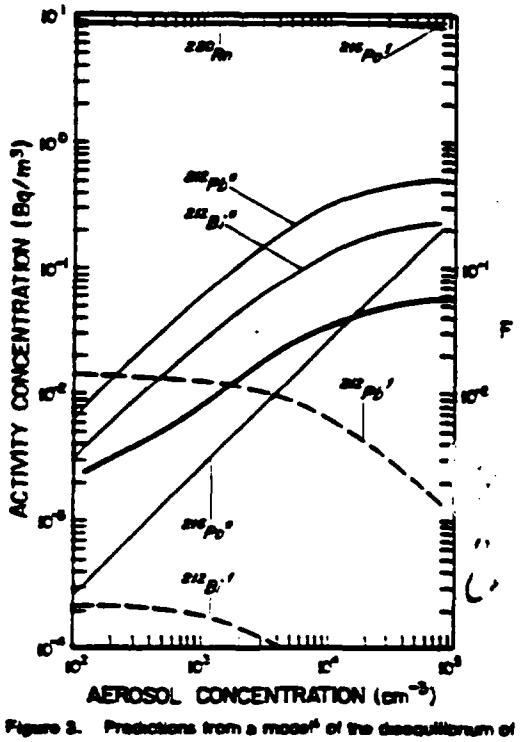

Figure 3. Predictions from a model[6] of the disequilibrium of thoron progeny in indoor air as a function of aerosol concentration for a ventilation rate of $\nu = 0.75\ h^{-1}$. Other parameters assumed in this representative calculation are a deposition rate to the walls of attached particles of $0.2\ h^{-1}$, a deposition rate to the walls for unattached particles of $40\ h^{-1}$, an aerosol attachment coefficient of $5 \times 10^{-2}\ cm^3\ h^{-1}$, and a thoron source term of $400\ Bq\ m^{-3}\ h^{-1}$. The superscript $a(f)$ refers to attached (unattached) particles and the scale at right refers to the thoron progeny equilibrium factor F.

Table I. The approximate ratio of the potential alpha particle energy for thoron progeny to radon progeny at various locations.[8,12]

Location	Approximate ratio
Italy (Latium), 50 dwellings	1.3
Canada (Elliott Lake), 95 dwellings	0.3
Hungary, 22 dwellings	0.5
Norway, 22 dwellings	0.5
Federal Republic of Germany (Western) 150 measurements	0.8
Federal Republic of Germany (Southwestern) 95 dwellings	0.5
France (Finistere), 219 homes	0.3
United States (20 states), 68 measurements	0.6
Hong Kong, 10 indoor sites	0.8

Equation 5 clearly shows a reduced, non-linear increase in PAEC due to thoron with PAEC due to radon. A second, and more important, response involves details of the dosimetry of radon and thoron. Although ^{212}Pb itself is not an alpha particle emitter, it is an important isotope in air controlling delivery of the alpha particle dose. ^{212}Pb has a half-life significantly longer than the longest important decay product of radon, ^{214}Pb, with a half-life of 1.6×10^3 s (27 min). Radioisotopes deposited in the bronchial tree have a probability of removal by mucociliary clearance before they can damage cells by decay, and due to its longer half-life, a larger fraction of ^{212}Pb gets removed. In addition, ^{212}Pb has a longer residence time in air and more opportunity to attach to aerosols and form larger particles. Larger particles are less effectively deposited on the critical bronchial epithelium of the respiratory tract and for this reason also ^{212}Pb delivers less dose for a given PAEC of the original air compared with ^{214}Pb.

The details of these dosimetry models are complicated and undergoing change, but at present the best estimates are that under typical atmospheric conditions, PAEC due to thoron is only one-third to one-fifth as effective as an equivalent PAEC due to radon in delivering dose to the lung.[14,15] Thus, in terms of overall health hazard, a 50 percent ratio for PAEC is reduced to an effective health effect of about 10 to 20 percent for thoron relative to radon. Here we have probably the major reason at present that thoron is not considered as important an indoor pollutant as is radon.

If attention is focused on dose to the whole population rather than to those individuals living under exposure conditions at the high end of the exposure distribution, effects of thoron are more significant using conventional dosimetric analysis. For example, in the United States it has been estimated that there are up to 20,000 deaths per year from lung cancer due to radon.[7,16] Applying the same underlying assumptions and analysis to thoron, one could scale the dose equivalence due to radon with the dose equivalence due to thoron to project that there are up to 4000 deaths per year due to indoor thoron. This is not an insignificant risk being comparable to other safety hazards of concern for the population such as accidental deaths due to poisoning or firearms and much greater than projections of the effects due to radioactivity from sources such as weapons testing fallout and emissions from power reactors. However, analysis such as this is complicated by not fully resolved societal issues of whether small risk spread over a large population warrants as much concern as higher risk to a smaller, more identifiable population.

Conclusions

There is much we do not know about thoron in the enviroment, and new developments could change the present situ-

ation, either toward more or less concern. It is worth pointing out that there exists little direct evidence for overall adverse effects of low exposure to either radon or thoron (on the order of less than 0.02 WL). For thoron progeny, there exist no epidemiological data at all, even for higher concentrations. The apparent developing consensus in the United States of the desirability of reducing further even low radon exposure is not without critics. The present guidelines for maximum acceptable radon levels depend on analysis of alpha particle radiation dose to the respiratory tract, and linear extrapolation of epidemiological evidence of increased lung cancer incidence at higher concentrations of PAEC due to radon progeny to these lower levels. Humans have evolved on an earth that has always had radon and thoron products in the air. Although in the minority, there are scientists who have studied this issue carefully who feel a threshold exists below which exposure to ionizing radiation poses no significant risk.[17] In fact, some scientists advance arguments why exposure to small amounts of ionizing radiation actually carries a beneficial effect. The arguments for the strongly ionizing alpha particle radiations of radon and thoron products do not seem as strong as the arguments for more weakly ionizing radiations (beta and gamma) but they nevertheless exist.[17,18] Such hormetic effects have been found to be true of a number of environmental agents from sunlight to certain trace metals in our diet.

Meanwhile, there are more straightforward questions about thoron that hopefully will be soon answered. One is whether, and to what extent, soil is the source for indoor thoron. This question is important should it become necessary to reduce thoron in houses by remedial measures. Does one reduce infiltration from the soil, as most often done at present for radon, or does one need to carry out other measures such as applying sealants to indoor surfaces of building materials? If it turns out that soil is indeed the major source of thoron, present mitigation programs reducing radon may automatically eliminate many potential problems due to thoron. A second important question is whether or not there exist regions or types of housing which possess unusually high levels of thoron. It was the discovery of such regions that gave much of the impetus to the present concerns with radon. Data on indoor thoron are at present too sketchy to answer this question but there are a number of studies underway that should provide information in the near future.

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Footnote #40

NCRP Report No. 76

RADIOLOGICAL ASSESSMENT: PREDICTING THE TRANSPORT, BIOACCUMULATION, AND UPTAKE BY MAN OF RADIONUCLIDES RELEASED TO THE ENVIRONMENT

Recommendations of the NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS

Issued March 15, 1984
First Reprinting May 1, 1985

National Council on Radiation Protection and Measurement
7910 WOODMONT AVENUE / BETHESDA, MD. 20814

NCRP No. 76

(1975) provides original data, as well as a summary of several other studies, on shielding provided by structures such as homes, office and industrial buildings and vehicles.

Shielding factors vary for a structure depending on whether the source of radiation exposure is ground deposited or is a cloud. The reported reduction factors (ratio of the exposure rate in the structure to the exposure rate in open air) for radionuclides deposited on the ground range from an average of about 0.27 for single story wooden houses to 0.06 for reinforced concrete and brick homes. Basements provide more protection than other areas of the house and average reduction factors range from 0.003 to 0.08. Large office buildings provide more shielding, and, therefore, lower reduction factors than homes (other than basements). For example, average reduction factors in one- and two-story office buildings range from 0.01 to 0.12.

Vehicles provide reduction factors in the range observed for one-story houses. The range for cars, pickups, buses, trucks and trains is 0.15 to 0.6. The range for trucks and trains is encompassed within the observed range for automobiles, pickups and buses.

In the case of cloud sources, only average reduction factors are available for buildings and vehicles. These values are probably representative of the upper range for a particular structure. If it is assumed that a range similar to that observed for ground deposits applies, then reduction factors might be as follows; wood frame homes, 0.6 to 0.9; masonry houses and masonry houses with basements, 0.3 to 0.6 and 0.2 to 0.4 respectively; and large office buildings, 0.05 to 0.2.

The distribution between these ranges for both ground deposited and cloud sources could be considered to be uniform because there is likely to be a continuous range of values due to difference in materials and architecture.

A precise analysis of reduction factors for individuals in a specific area would require information on the percentage of the different type dwellings in the area. Data on the numbers of one- and two-story single family dwellings in different regions of the United States are given for 1970 by Moeller and Underhill (1976).

For individuals, it is also necessary to determine how much time is spent at home (in one- or two-story wood or masonry dwellings) and how much time is spent in large office or industrial buildings; a general range of 6 to 10 hours per day or 30 to 50 hours per week can be assumed. Seventy to 95% of a person's time spent indoors is the equivalent of 118 to 160 hours per week, of which 30 to 60 hours would be in office or industrial buildings.

These time distributions and ranges can be used in conjunction with the range of reduction factors of Burson and Profio (1975) to estimate the range of effective reduction factors expected for individuals.

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NCRP REPORT No. 94

Exposure of the Population in the United States and Canada from Natural Background Radiation

Recommendations of the NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS

Issued December 30, 1987

National Council on Radiation Protection and Measurements
7910 WOODMONT AVENUE/Bethesda, MD 20814

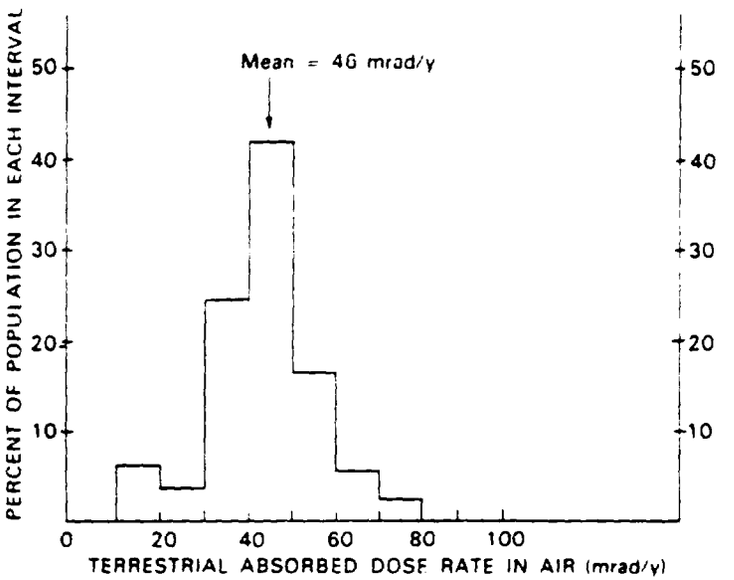

Fig. 5.7. Terrestrial absorbed dose rate in air measured by aerial surveys over nuclear power plant sites. Rates in mrad/y may be converted to mGy/y by dividing by 100.

outdoors and 230 ± 140 μSv/y (23 ± 14 mrem/y) indoors are adopted for this report.

5.4 Indoor Exposures

Exposure to terrestrial gamma radiation indoors is modified by the materials of construction and by the position of the individual within the structure. Wood plastic, metal and glass have relatively little activity, while brick, concrete and other masonry tend to be similar to soil in the surrounding area.

Walls provide some shielding from outdoor radiation. Cameras and Rickards (1973) measured a factor of 2 reduction with a 12.5 cm wood frame wall and a factor of 4 with a brick wall. A 20 cm concrete wall gave a reduction factor of 20. They also indicated that windows and doors provided little shielding. Roof shielding against sky-shine is probably not important, since the unshielded component from this source is only a few percent of the total terrestrial radiation. Wooden houses can also reduce exposure by separation from the source. Exposures on upper floors of wooden frame houses are usually lower than the ground floor.

On the other hand, a person indoors is completely surrounded by sources (4-pi steradians geometrically) which should increase expo-

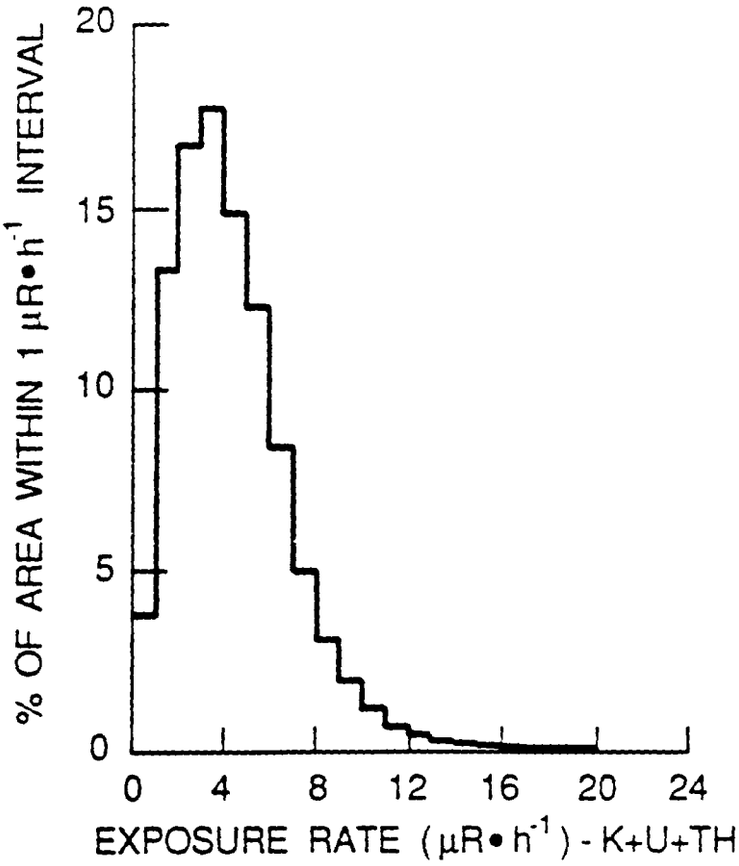

Fig. 5.9. Distribution of terrestrial gamma-ray exposures in Canada from airborne measurements. Note that the exposures shown should be increased by 10%. Rates in μR/h may be converted to μGy/h by dividing by 115.

the much greater number of outdoor measurements in assessing population exposure. This factor has no scientific basis but cannot be completely replaced until adequate indoor data become available.

UNSCEAR, in its reports through 1982, has used a value of 1.2 for the indoor/outdoor ratio based almost entirely on measurements in European masonry houses. Oakley (1972) developed a "housing factor" of 0.8 which was considered to be more representative of the frame houses assumed to dominate housing in the United States. This factor was used in the 1975 NCRP report on natural background radiation.

The measurements of indoor vs. outdoor exposures from NCRP (1975a) are listed in Table 5.5. These are generally confirmed by the more recent measurements of Miller (1986) who collected data for 80 houses in New York State and the New York City metropolitan area. He found that outdoor absorbed dose rates in air averaged 350 μGy/y (35 mrad/y), while the rates were 240 μGy/y (24 mrad/y) inside 60

TABLE 5.5 - *Comparison of indoor to outdoor absorbed dose rates in air*[a]

Building materials (outer walls)	Percent of outdoor absorbed dose rate in air	Remarks	Reference
Frame, brick, and stone apartments and houses	Approx. 80–100	17 dwellings	Solon *et al.* (1960)
Mostly wood frame	70	160 single homes	Lowder and Condon (1965)
Frame	82	5 single homes, 1st floor	Yeates *et al.* (1970)
Brick	96	1 apartment, 2nd floor	
Steel and concrete	87 106	4 office buildings	
Mostly wood frame	75	110 single homes	Lindeken *et al.* (1971)

[a] Oakley (1972) compiled the table in terms of dose equivalent. The same ratios would hold for absorbed dose in air.

frame houses and 400 μGy/y (40 mrad/y) inside 20 masonry houses. Grasty *et al.* (1984) have a useful discussion of indoor exposure considerations.

There are marked differences between wood-frame and masonry houses which modify actual population exposure. The Statistical Abstract of the United States (USBC, 1986) indicates that the 92 million year-round dwellings in the U. S. (1983) are about two-thirds single family and 15 percent apartments with 5 or more units. The latter should be largely masonry, while the smaller buildings are apparently about 50 percent masonry or masonry-faced and 50 percent wood frame with no masonry in the walls (USBC, 1971). About 50 percent of single-family houses have masonry basements, and any time spent in the basement is equivalent to time in a full masonry structure.

Moschandreas (1981) has estimated that about 20 percent of the day is spent outdoors. In addition, the amount of time spent at home is modified by occupation. The Statistical Abstract (USBC, 1986) shows the approximate division of population over 16 as 60 percent employed, 6 percent full-time students, 15 percent full-time housekeepers, and 19 percent other, including unemployed. The one-quarter of the population below 16 include the 7 percent of the total population of pre-school age while most of the rest under 16 should be additional full-time students. Thus, probably two-thirds of the population spend six or more daytime hours at work or school in masonry structures.

In view of these modifying factors, it would seem that attempting to correct for type of dwelling would not make a significant change in the population exposure, although it could for an individual. Thus, in this report, the average indoor exposure will be assumed as equal to that measured outdoors for the U.S. The Canadian estimate was presented in Section 5.3. It is hoped that actual survey data for indoor exposures will become available in the future.

5.5 Variations in Terrestrial Radiation

There are locations where the levels of external radiation from natural sources are markedly elevated above the average. A few of these will be considered in Section 8 but it must be realized that there are many more that have not yet been identified. This section will consider only the sources of variation in average exposures.

In general, the variability is not large. For instance, combining the available outdoor data for Italy, Japan, West Germany and the United States, as given in UNSCEAR (1982), gave a single normal distribu-

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SEARCH FOR BUILDING MATERIALS AS SOURCES OF ELEVATED RADIATION DOSE

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Abstract—A systematic approach to finding materials that cause elevated γ radiation exposure rates or ^{222}Rn progeny working levels in buildings was tested in the Atlanta area. With the proposed procedure, exposure rates in planned structures would be derived from a radiation survey of the material performed with a NaI(Tl) detector. Working level values under specified conditions would be inferred from an additional measurement of ^{226}Ra concentration in the material. To quantify the procedure, surveyed building materials were analyzed for radionuclide content and categorized according to an exposure rate index related both to the survey meter count rate and the radionuclide concentration. An available calculational model was used to predict from the index the exposure rate in a room totally enclosed by radioactive material. An additional model was developed for application to structures where the radioactive material is only in the floor or walls. ^{222}Radon concentrations in an enclosed space were also predicted according to an available model. Exposure rates were measured in newly constructed buildings to test the approach. Measurements in older buildings found elevated γ-ray emission from concrete blocks made with phosphate slag from a phosphorus producer in Alabama. Buildings with walls of these blocks were used to test the prediction of ^{222}Rn concentrations in building air due to ^{226}Ra in construction material. The calculational models predicted that the ^{226}Ra concentration of approx. 20 pCi/g in these blocks would result in increases of the average exposure rates by 10 μR/hr and of ^{222}Rn concentrations by 0.2 pCi/l. under specified conditions. Observed levels were consistent with these predictions, but the major fraction of ^{222}Rn in room air was attributed to inflow from the ground beneath the building.

INTRODUCTION

BUILDINGS that contain unusually high ^{226}Ra levels in walls or floors have been found in several locations in the U.S. There is concern that some of the buildings will cause excessive radiation doses to the total body due to γ rays emitted by the ^{214}Pb and ^{214}Bi progeny of ^{226}Ra, and to the lungs due to α particles emitted by ^{218}Po and ^{214}Po progeny formed through the decay of gaseous ^{222}Rn in air. The ^{232}Th decay chain and ^{40}K also contribute to total body radiation dose, and ^{220}Rn progeny, to the lung dose. The U.S. Environmental Protection Agency has proposed regulations under the Resource Conservation and Recovery Act of 1976 that prohibit building products manufactured from certain wastes if these will raise levels by 5 μR/hr or 0.03 WL above the usual radiation background. Although the proposals have not been implemented and guidelines with different criteria are being followed in various circumstances, one can anticipate that extensive surveys will be undertaken to identify building materials that may cause unacceptably high radiation doses. A systematic approach to such surveys is described here.

Radiation detection devices in current use detect elevation in γ-ray exposure rates by as little as 1 or 2 μR/hr relative to the natural terrestrial plus cosmic radiation background, which in the U.S. is generally between 5 and 25 μR/hr. For spot (i.e. brief) measurements during extensive surveys of homes, pres-

surized ionization chambers (Ni80) and plastic scintillators (Ko78a) have been successfully utilized, while thermoluminescent dosimeters have served for long-term measurements (Ni80; Mj80). In determining the increment due to building material, consideration needs to be given to (1) procedures for predicting the radiation exposure potential of building materials before the building is constructed; (2) defining the natural background within 1 or 2 μR/hr when it ranges over 5 or even 10 μR/hr at some survey areas (NCRP75); and accounting for fluctuations in the background within buildings by 1 or 2 μR/hr (Li73; Ni80). Two calculational approaches (Kr71; Ko78) have been proposed to deal with the first item; the second and third items would have to be resolved by more detailed measurements when the exposure rate elevation is near a regulatory limit or action level.

Determining the WL value due to building material is a much more difficult task. Unlike γ radiation in structures, ^{222}Rn progeny in air fluctuate widely in concentration and are not easily traceable to their source with a survey instrument. Although WL values can be measured below 0.03 WL by approx. 2 orders of magnitude with various available detection devices, an elevated value is not necessarily due to man-made conditions and a low value does not eliminate the potential for elevated values at other times. At the present state of knowledge, if elevated WL values due to building material are sought or are believed to occur, a study must be performed to distinguish among the various sources and factors that lead to high values.

Elevated ^{222}Rn concentrations in building air are currently believed to come mainly from the ground or fill beneath the building. Concentrations in building air fluctuate with changes in the infiltration (or ventilating) rate due to such common activities as opening doors and windows, and with changes in atmospheric and soil conditions that affect the radon flux into the building in ways not yet well defined. Fluctuations in the ^{222}Rn concentration in outside air and utilization of water from groundwater supplies high in ^{222}Rn add to the variability in building air (OECD79). The ratio of WL value to ^{222}Rn concentration is affected by the degree to which the ^{222}Rn progeny are removed from building air by filtration and surface deposition. At equilibrium, a value of 0.03 WL is associated with a ^{222}Rn concentration of 3 pCi/l.; typical values indoors correspond to approx. 40% of equilibrium (UN77). At that level, a ^{222}Rn concentration of 7.5 pCi/l. results in 0.03 WL.

Surveys in the U.S. show averages in homes between 0.003 and 0.007 WL, with distributions that are generally log-normal and include values well above 0.01 WL (Al74; Pe77; Gu79; Ge80). Very high ^{222}Rn concentrations—more than 10 pCi/l. in some cases—have been found in some homes (Ru79; Go81).

In the U.S., concrete made with slag from the production of phosphorus that contains elevated levels of ^{226}Ra has been found in Idaho (Bo77; Pe78; US80) and Montana (Li78; US80) buildings. The buildings were identified by their elevated γ radiation exposure rates. ^{222}Radon and WL values were in the normal range but included some values above 0.03 WL (Pe78; Ll78). No distinction was made concerning the source of ^{222}Rn for building air above 0.03 WL. In Europe, similarly elevated levels were found by γ radiation exposure rate measurements for plasterboard made with phosphogypsum and aerated concrete with alum shale (OECD79). Swedish houses with alum-shale-based aerated concrete have been found with ^{222}Rn and WL values as high as 11 pCi/l. and 0.04 WL, respectively (Sw80). Other homes, however, have similarly high values despite building materials of lesser ^{226}Ra content due to low ventilation rates (0.01–0.3 per hr, compared to average values of 0.6 to 0.7 per hr in Sweden).

The following approach to identifying materials leading to elevated radiation exposure rates and WL values was followed in this study:

(1) Find materials that in bulk show higher than normal γ-ray exposure rates with survey meters.

(2) Analyze these materials for radionuclide content.

(3) Compute a radiation exposure rate index from the radionuclide concentrations and compute the ^{222}Rn flux density from the ^{226}Ra concentration.

(4) Compute the γ-ray exposure rate and ^{222}Rn concentration as a function of building dimensions from the index and flux density, respectively.

(5) Compare these with exposure rates and ^{222}Rn concentrations from material with average radionuclide (including ^{226}Ra) concentrations.

In existing buildings with elevated radiation exposure rates or WL values, the material (if any) that emits more γ-rays is identified by spot survey and the ^{222}Rn flux density from the material is measured (Co76). To determine whether elevated exposure rates indicate potentially high ^{222}Rn concentrations in building air, the concentration of ^{226}Ra relative to ^{228}Ra and ^{40}K is obtained by analyzing a sample of the material or by on-site spectral analysis with a portable γ-ray spectrometer.

Construction materials that were potential sources of elevated radiation exposures—mostly brick and concrete—were surveyed in the field and sampled for radionuclide analysis. Calculational models were then applied to predicting γ-ray exposure rates and ^{222}Rn concentrations in buildings. A model was developed for a situation to which the available models did not apply. Newly constructed homes in several subdivisions were surveyed to compare measured with predicted exposure rates. Older structures were surveyed to search for material with elevated radionuclide levels, and this material was traced to its source. Finally, ^{222}Rn concentrations in air at selected buildings constructed with the high-^{226}Ra material were measured to compare with predicted values.

PROCEDURE

Building materials in the yards of major suppliers in the Atlanta area were surveyed to determine the external radiation exposure potential of the materials (Ka79; Ei80). The materials were surveyed in large piles, usually 1–2 m from side to side, with a NaI(Tl) detector and rate meter. The detector was cylindrical, 5 cm in diameter × 5 cm long. The low-energy discriminator was set to detect only γ-rays above 0.09 MeV. The detector was calibrated relative to a pressurized ionization chamber (PIC) for mixed terrestrial radiation. For measurements, the detector was placed directly on the pile of materials. The natural radiation background and the influence of adjoining materials were determined by moving the detector in various directions from the pile.

Samples of many of the materials were analyzed in the laboratory for ^{226}Ra, ^{228}Ra and ^{40}K with a Ge(Li) detector and spectrometer. Radium-226 was determined by measuring the 352- and 609-keV γ-rays of ^{214}Pb and ^{214}Bi, respectively. For confirmation, its 186-keV γ-ray was measured and it was assumed that 58% of the γ-ray was due to ^{226}Ra (0.035 γ-ray per disintegration) and 42% from ^{235}U (0.54 γ-ray per disintegration, 0.0466 pCi ^{235}U/pCi ^{238}U). Radium-228 was determined by measuring ^{228}Ac and ^{208}Tl γ-rays and assuming that these were in equilibrium with ^{228}Ra. For ^{40}K, the 1.461-keV γ-ray was measured.

A group of 23 new homes, built with these materials, was surveyed with the PIC to measure the exposure rate; 6 older homes and 29 multi-unit structures were also surveyed. The PIC is a steel sphere, 0.3 m in diameter, filled with argon at a pressure of 25 atmospheres. It was calibrated in terms of μR/hr with a ^{226}Ra source checked by NBS. The PIC was taken to various locations in houses, including the centers of rooms and near walls, to measure typical values and to search for sources of elevated exposure rates. Where materials responsible for higher radiation exposure rates were found, the radionuclides were identified by analyzing a sample of material or by operating a portable Ge(Li) detector with spectrometer nearby.

A computer calculation of the exposure rate in a room with radioactive walls or floors was performed with the modified code FUDGE 4 (Ma64). The values of μR/hr at a grid of locations within a room for the three terrestrial radionuclides at unit concentrations in walls and floor were obtained for square structures, 2.4 m high with various floor areas between 10 and 150 m^2. Wall and

floor surface densities were 17 and 23 g/cm^2 respectively. The values are approximate because only mean values of γ-ray energies and fractions were used for the two decay chains (^{238}U: 0.81 MeV, 210%; ^{232}Th: 0.88 MeV, 270%; ^{40}K: 1.46 MeV, 10.7%). Results depend noticeably on the choice of buildup factors, for which Taylor's sum of exponentials was used (Mo73), and were clearly inappropriate near surfaces.

The component of concrete blocks responsible for the elevated exposure rate was identified by inquiries of suppliers on the basis of date of construction and physical characteristics. Upon being contacted, the supplier of that component provided all requested information concerning the source and its extent, and also samples for analysis.

Radon-222 concentration in air was measured in 125-cm^3 radon scintillation (Lucas) cells. The samples were collected in three office/laboratory buildings at the Georgia Institute of Technology during the mornings for 1-min periods by admitting air into evacuated cells (Ka83). Concrete blocks with considerably elevated ^{226}Ra concentrations were used extensively in constructing two of these buildings. Measurements were performed to determine differences in air between outside and inside, among different floors within a building, among buildings, and as functions of season, time and atmospheric conditions. Radon flux density was measured by collecting ^{222}Rn on 200 cm^3 charcoal in 80-cm^2 containers sealed to the ground or to floors and walls for 3 days and analyzing the charcoal by γ-ray spectrometer. The ventilation rate in buildings was determined by measuring SF_6 tracer with a gas chromatograph. Radon analyses of water showed that water use was not a source of ^{222}Rn in building air.

RESULTS AND DISCUSSION

The radium concentrations in the brick and concrete samples listed in Table 1 were observed to be in the four distinct categories labelled "very low," "low," "medium" and

Table 1. Average radionuclide concentration and exposure rate potential of construction materials

Material	No. of Samples	Radionuclide concentration, pCi/g ^{226}Ra	^{228}Ra	^{40}K	Exposure rate index, uR/hr	Category
Brick	2	0.2	0.08	0.3	0.7	very low
	5	0.6	0.9	12.	5.8	low
	24	1.8	1.9	17.	11.8	medium
	6	3.5	2.9	27.	19.7	high
Concrete, poured and block	3	0.7	0.5	8.0	4.1	low
	14	1.4	1.7	26.	12.1	medium
Tile	9	1.9	1.0	5.3	7.4	low
	1	1.7	2.1	28.	14.2	medium
Wood	1	0.04	0.013	0.3	0.2	very low
	5	--	--	--	--	very low
Wall ceiling board and tiles	5	--	--	--	--	very low
Glass panes	2	--	--	--	--	medium
Glass fiber insulation	3	--	--	--	--	medium
Roofing	1	--	--	--	--	low
Metal braces and wires	2	--	--	--	--	low

"high." It should be noted that many more brick and concrete piles were in the "medium" category than indicated by the numbers of samples because preference was given to analyzing samples from piles that caused unusually high or low count rates. The average for each was separated by approximately a factor of two from the other. In only a few instances (Ei80) did these clear distinctions among categories not apply. The ^{226}Ra and ^{228}Ra concentrations in the "medium" category were somewhat high in relation to surface soil from 16 locations throughout Atlanta, in which the range was from 0.3 to 2.6 pCi/g and the average was 1.1 pCi/g (Ka79).

An index, I, in μR/hr, was selected to estimate the radiation exposure rate potential due to the combined radionuclide concentrations, C, in pCi/g, of the ^{238}U progeny, ^{232}Th progeny and ^{40}K in a material. This index was taken to be the exposure rate measured 1 m above an infinite hemisphere or slab with respect to the γ-rays emitted by the radionuclides. The following relation, with the first coefficient modified on the basis of more recent decay information, is given by Beck (Be72; Be79):

$$I = 1.90\, C_{^{238}U} + 2.82\, C_{^{232}Th} + 0.179\, C_{^{40}K}. \quad (1)$$

Concentrations of ^{226}Ra and ^{228}Ra can be substituted for those of ^{238}U and ^{232}Th, respectively, because X- and γ-rays in the chains before radium contribute little to the total exposure rate. Krisiuk *et al.* first used this approach (Kr71) in terms of an infinite sphere, i.e. approx. twice the above exposure rate per concentration. Their coefficients were approx. 20% higher than double those in equation (1) for ^{238}U and ^{232}Th, possibly because they used different γ-ray intensities.

The average indices given in Table 1 maintain the distinct categories observed for radium concentrations in brick and concrete. Extreme values in a category were generally within 20% of the average (Ei80). These indices compare to measured terrestrial background exposure rates between 2 and 12 μR/hr in the Atlanta area (Ka79). In addition, cosmic radiation in Atlanta (elevation 300 m) contributes 3.8 μR/hr (NCRP75; Li72) to the outside exposure rate and possibly 10% less inside (NCRP75).

The survey results with the NaI(Tl) detector placed above large piles of materials were consistent with the index values given in Table 1. The count rates approximated the exposure rate indices for terrestrial radiation on the basis of the calibration factor for the survey meter of 600 count/min per μR/hr because the detector is relatively insensitive to cosmic radiation. The contribution to the count rate from widely varying levels of terrestrial radiation had to be taken into account, however, as well as radiation from nearby stacks of other materials. The tests with concrete, brick, tiles, and wood gave sufficient confidence to apply the method to the other materials listed in Table 1, as long as these materials were not in the "high" radiation category.

The building survey results in the first three rows of Table 2 show wide ranges of

Table 2. Exposure rates in structures

Buildings	No.	Average exposure rate and range, μR/hr outside	inside typical	inside low	inside high
New homes	23	6.3-12	7.1-15	6.7	19
Older homes	6	6.5-9.7	8.0-16	6.9	18
Multi-unit structures*	24	9.0-16	8.0-18	6.3	21
Multi-unit structures, with phosphate slag concrete block	5	7.8-16	28-34**	26**	48

* Apartment buildings, offices, schools and warehouses.
** Refers to areas with these blocks; exposure rates in other areas were similar to those in other buildings.

exposure rates both inside and outside. Inside exposure rates were consistently higher, although generally by small increments, in buildings constructed with brick or concrete. The extent of elevation depended on the materials that were used. In the one wooden frame house that was surveyed, the exposure rate was lower than outside (Ka79).

Within buildings, the radiation exposure rate varied with the distance from the main source of radiation. The highest values shown in Table 2 were measured near the surfaces of masonry walls or floors, particularly in the basement. Exposure rates decreased from basement to ground floor to second floor, and were lowest away from concrete or brick on upper floors (Ei80).

Extensive measurements indicate that inside γ-ray exposure rates, excluding the cosmic ray contribution, in homes with masonry walls are typically 1.2–1.6 times as high as the outside terrestrial exposure rates (Ko78; Ni80). Moeller and Underhill (Mo76) suggest a factor of 1.3 for the ground floor relative to outside, and factors 30% higher in the basement and 15% lower on the second floor. In frame houses, on the other hand, walls contribute fewer γ-rays than they absorb, resulting in inside exposure rates that are 0.7–1.0 of outside values (Ni80; Mo76).

Estimation of the indoor radiation exposure from construction material with the index I from equation (1) appears feasible for relatively simple situations. Krisiuk *et al.* (Kr71) arrive at an indoor γ-ray exposure rate equivalent to $2\,I$ if walls, floor, and ceiling are infinitely thick with regard to the emitted γ-rays. This is the upper limit of the indoor exposure rate due to the material. It can be encountered in buildings with very thick or multiple walls. Koblinger (Ko78), in calculating the γ-ray exposure rate in a 4- × 5- × 2.8-m room from 20-cm-thick concrete floors, wall and ceiling, obtained approx. 1.8 I. Extrapolation of his values to 10-cm-thick concrete yields 1.4 I. Calculations for a square room or building with masonry walls or floor resulted in the radiation exposure rates in terms of I shown in Figs. 1 and 2 for ^{238}U. Similar curves were obtained for ^{232}Th and ^{40}K. The values are lower than the Koblinger results extrapolated to 10-cm-thick material because a nonradioactive ceiling had been postulated. This is generally applicable to 1-family homes in the U.S.

Estimates of indoor γ radiation exposures must include the terrestrial exposure rate attenuated by walls and floors and the slightly attenuated cosmic ray exposure rate. For example, in computing the exposure rate on the first floor of a 12- × 12-m 1-family house in Atlanta, the following sources must be considered if the terrestrial exposure rate is 6 μR/hr and the building material index is 6 μR/hr: terrestrial radiation 6 μR/hr × 0.3 (Ka79), cosmic rays, 3.8 μR/hr × 0.9, and building materials 6 × 0.8 (see Fig. 1), for a total of 10 μR/hr. If the I value for the material were 12 μR/hr, more typical of the index for concrete and brick at Atlanta, the indoor levels would be 15 μR/hr. The sample calculations yield reasonable results compared to the measured values but suggest the difficulty of calculating precise exposure rates from a relatively simple model.

In the five structures referred to in the last row of Table 2, much higher radiation exposure rates than usual were found near the surface of concrete block walls. These walls were in two office/laboratory buildings at the Georgia Institute of Technology, two warehouses, and in portions of corridors in the former Atlanta airport terminal. The buildings were among 29 multi-unit structures randomly selected for this test survey. The structures or parts of structures in which the walls had higher exposure rates were built in the mid-1960s.

Samples of block from three of these buildings were available for analysis. The blocks had the following averages and ranges of radionuclide concentrations (Ka83):

^{226}Ra	19	(18–21) pCi/g
^{228}Ra	1.0	(0.9–1.2)
^{40}K	9.1	(7.4–9.9)

The ^{226}Ra concentrations are extraordinarily high for building material, whereas ^{228}Ra and ^{40}K are in the "low" category of Table 1. The

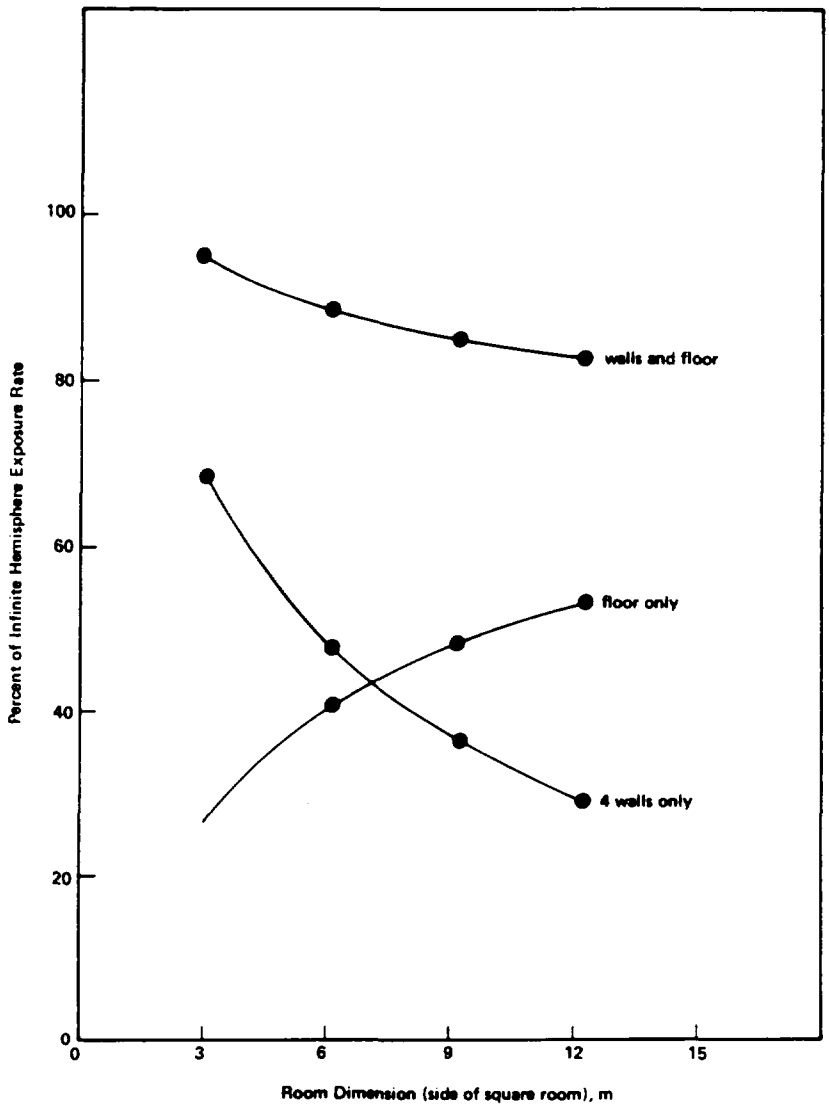

FIG. 1. Average γ-ray exposure rate index within building due to ^{238}U chain in walls and floor.

sample of brick or concrete measured in this study.

A γ-ray spectrum obtained on site with a portable Ge(Li) detector showed the much higher ^{226}Ra concentration in the concrete blocks relative to ^{228}Ra:

	Relative amount, pCi		
	^{226}Ra	^{228}Ra	^{40}K
Building with concrete blocks	10.0	1.0	8.7
Control building	1.1	1.0	7.7

Absolute calibration is difficult because of the different concentrations of the three radionuclides in the high-^{226}Ra walls and in the nearby floor and ceiling with normal ^{226}Ra levels. Relative calibration (approximately a straight line of energy vs efficiency on a log–log graph) is feasible on the basis of numerous γ-rays of known intensity from either ^{226}Ra or ^{228}Ra progeny. The relative ^{226}Ra amounts in the building with concrete blocks are intermediate to the concentrations in these blocks and in ordinary concrete.

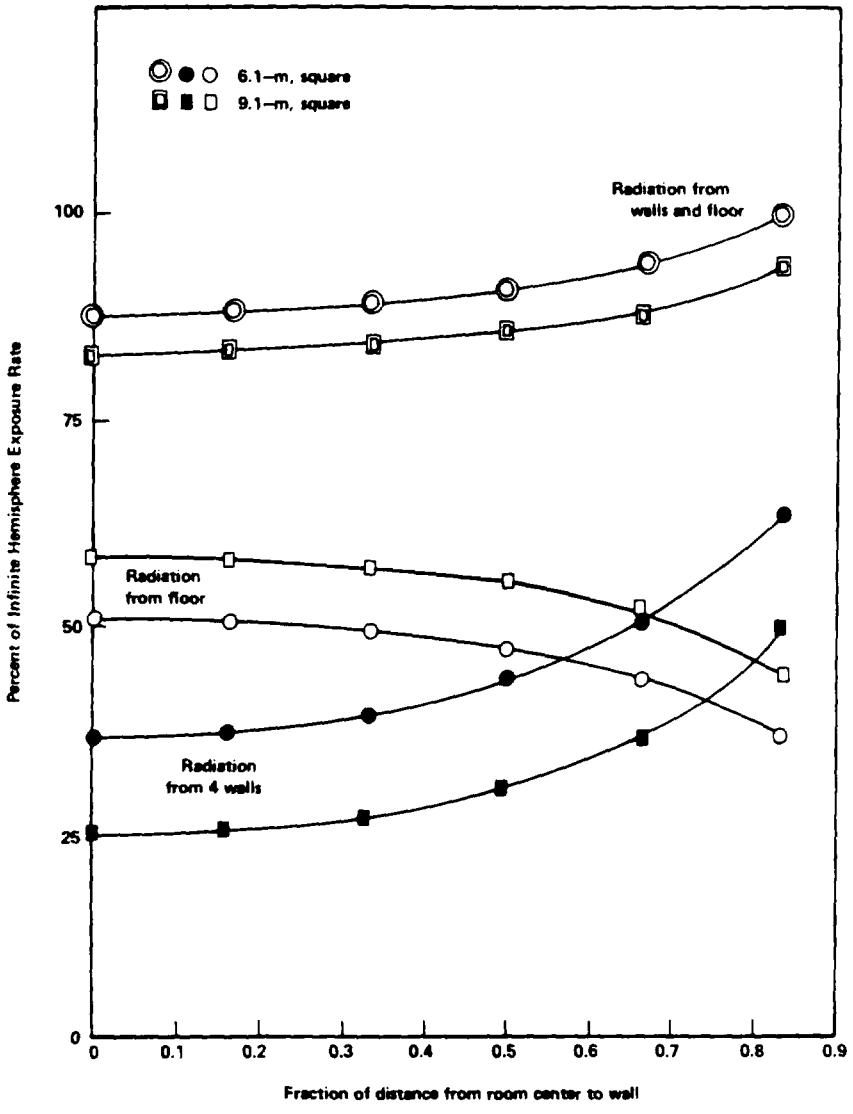

FIG. 2. Variation within building of γ-ray exposure rate index due to ^{238}U chain in walls and floor (height above floor: 1.2 m).

The source of the elevated ^{226}Ra levels in the blocks was found to be slag from the thermal reduction process for phosphorus production at a plant in Muscle Shoals, AL. Five samples of slag provided by the plant contained the following averages and ranges of radionuclide concentrations:

^{226}Ra	28	(19–38)	pCi/g
^{228}Ra	0.8	(0.6–0.8)	,,
^{40}K	4.2	(1.9–5.7)	,,

The plant uses a mixture of phosphate ore from Florida and Tennessee, typically 33% of the former at an average ^{226}Ra concentration of 65 pCi/g and 67% of the latter at 4 pCi/g (Wi75).

Until 1969, most of the slag was prepared in an expanded form; thereafter it was in a quenched form until all sales were halted in 1978 (Ma78). Both were used extensively for manufacturing concrete blocks: the expanded form, to make lightweight blocks that contain approx. 80% slag, and the quenched form in

blocks that contain 30–40% slag. The ^{226}Ra concentration due to slag in the lightweight block would be expected to be 22 pCi/g, only slightly higher than in this study. The blocks with quenched slag would contain 8–11 pCi/g, consistent with concentrations of 8 pCi/g found in blocks produced in the early 1970s (Wi75). According to the producer, approx. 2.8 million tons of slag were shipped mostly to construction and building material companies in Alabama, Tennessee, Mississippi and Georgia between 1953 and 1978. Of this amount, 380,000 tons were shipped to two firms in the Atlanta area between 1962 and 1966 (Ma78). This slag tonnage corresponds to 480,000 tons of lightweight concrete blocks, which may have been used to construct several thousand buildings.

Measurements of ^{222}Rn concentrations at two buildings (NE and ER) in which inside walls were constructed with numerous high-^{226}Ra concrete blocks and at a control building (CE) are consistent with previous observations (St80): ^{222}Rn inside were much higher than outside and concentrations on the ground floor were higher than on upper floors (Ge80). Taken together, the data in Table 3 suggest that the ground beneath contributes a major fraction of the ^{222}Rn inside the building and that ^{222}Rn accumulates in the building due to limited air exchange with the outside.

Although ^{222}Rn concentrations in air at the two test buildings were much higher than in the control building, calculations based on flux density and ventilation rate measurements (Ka83) lead to the conclusion that the blocks with high ^{226}Ra contents are not a major source of ^{222}Rn. The results summarized in Table 4 are based on ^{222}Rn flux density measurements in soil just ouside the buildings and on concrete floors and walls (both high-radium blocks and poured concrete at normal ^{226}Ra levels). Measured ventilation rates showed an average exchange of inside with outside air of 1.1 per hr in all three buildings. Because measurements on the slab at the lowest floor of the buildings showed much higher ^{222}Rn flux densities over

Table 3. Average morning ^{222}Rn concentrations in building and outside air

Location	^{222}Rn concentration, pCi/L				
	Outside	Basement	1st floor	2nd floor	3rd floor
CE building	0.1 (6)*	0.7 (8)	0.4 (2)	0.5 (4)	0.3 (2)
	<0.1-0.3	0.2-1.9	0.4-0.5	0.1-1.0	<0.1-0.6
NE building	0.2 (11)	---	1.3 (11)	0.7 (10)	0.8 (4)
	<0.1-0.4		0.9-1.8	0.5-1.0	0.6-0.9
ER building	0.2 (5)	2.4 (7)	2.0 (5)	1.8 (4)	---
	<0.1-0.3	0.4-3.7	0.1-3.2	0.9-2.2	

* Number of measurements in parentheses; range of values shown on second line.

Table 4. ^{222}Rn concentrations in building air calculated from ^{222}Rn flux density and outside air measurements

Location	^{222}Rn concentration from source, pCi/L				
	Ground	Unbroken slab	Concrete walls and floors*		Outside air
CE building	0.2	(0.04)	0.06	---	0.1
NE building	0.6	(0.06)	0.05	0.1	0.2
ER building	1.1	(0.04)	0.03	0.2	0.2

* the first value is for materials with normal ^{226}Ra, the second is for materials with high ^{226}Ra

cracks and seams than on the unbroken floor, it is believed that ^{222}Rn from the ground enters buildings mostly at such breaks and at openings for pipes and ducts. Hence, concentrations based on flux densities measured at the ground (see Table 4) appear to be more appropriate for predicting ^{222}Rn levels in indoor air than concentrations based on measurements at the unbroken slab.

Measurements of WL values in basements of 17 houses in Alabama that were built with phosphate-slag-bearing concrete blocks and in five control basements (US80; Ma78) confirm the need to distinguish among sources of ^{222}Rn before instituting control actions. Although the averages of 0.018 and 0.014 WL, respectively, for the test and the control values were not significantly different, some test homes had relatively high levels that in the absence of specific information might be attributed to the building material.

The values in Table 5 were calculated to compare ^{222}Rn concentrations in building air and γ-ray exposure rates expected from concrete with a medium exposure rate index and from concrete blocks similar to those found in Atlanta. In addition to the indicated room dimensions, ^{226}Ra concentrations and radiation exposure rate indices, the following values were assumed: concrete blocks are 0.14 m thick and have an effective density of 1.0 g/cm^3 (the concrete density is actually 2.3 g/cm^3, but the blocks are hollow); their emanation fraction is 0.05 (an average value of 0.03 was determined) (Ka83), and the ventilation rate is 1 room volume per hr. The γ-radiation exposure rate was computed on the basis of exposure rate indices from Fig. 1 for all rows in Table 5 except 4 and 6, for which 1.4 I was used. Concentrations of ^{222}Rn were computed for equilibrium among exhalation from radium-bearing material, air turnover, and radioactive decay (Ka83).

These estimates suggest that the presence in walls of the high-^{226}Ra concrete blocks increases the average γ-ray exposure rate within the rooms used in the examples by approx. 10 μR/hr, but the ^{222}Rn concentration in building air by only 0.2–0.3 pCi/l. The situation in the fifth row is comparable to the ER and NE buildings. In these two buildings, the predicted exposure rate, with 5 μR/hr

Table 5. Estimated radiation exposure rates and ^{222}Rn concentrations in air from building material

Structure	Radiation exposure index of material, μR/hr	Exposure rate, μR/hr	^{222}Rn concentration, pCi/l
basement, 1-family house 12 x 12 x 2.4m	12	9.9	0.034
	12 (floor) and 44 (walls)	19.2	0.20
	44	36.1	0.42
multi-story masonry building 9 x 9 x 3.0m	12	16.8	0.049
	12 (floor and ceiling) and 44 (walls)	27.3	0.28
	44	61.6	0.61

Notes: 1. I = 12 μR/hr corresponds to $C_{Ra\text{-}226}$ = 1.7 pCi/g; for I = 44 μR/hr, $C_{Ra\text{-}226}$ = 21 pCi/g

2. Exposure rates do not include values due to cosmic rays or the ground; Rn-222 concentrations do not include values from outside air, the ground, or gas and water supplies.

3. Exposure rates in the home and in the office with mixed materials depend on location in room; these are average values based on Figure 1.

added due to radiation from cosmic rays and the ground outside, corresponded to average measured values. The predicted ^{222}Rn increment of 0.2 pCi/l could not be attributed to building materials on the basis of measurements alone because of the much greater fluctuations in the totals shown in Table 3.

SUMMARY AND CONCLUSIONS

A survey in Atlanta found that the only building materials that resulted in γ radiation exposures comparable to terrestrial levels were concrete and brick. Typical radionuclide concentrations in these two materials were approx. 1.6 pCi/g for ^{226}Ra, 1.8 pCi/g for ^{228}Ra and 20 pCi/g for ^{40}K. Some groups of materials with considerably lower and higher radionuclide concentrations were also found. Surveys in buildings showed γ radiation exposure rates ranging from 7 to 18 μR/hr compared to 6–16 μR/hr outside.

In a few buildings, inside walls had been constructed with concrete blocks that caused exposure rates near 40 μR/hr. These blocks had been made with slag from a phosphorus production plant that utilized phosphate minerals high in ^{226}Ra content. The blocks contain ^{226}Ra at a concentration of 19 pCi/g.

Measurements in two buildings with high-^{226}Ra blocks yielded average ^{222}Rn concentrations in air of 2.2 and 1.0 pCi/l., compared to 0.6 pCi/l. in a control building. Measurements of ^{222}Rn flux density at walls and the ground suggested, however, that these differences were due mainly to different ^{222}Rn exhalation rates from the ground.

A radiation exposure rate index I, in units of μR/hr, is proposed for relating radionuclide concentrations in building materials to γ-ray exposure rates over blocks of the material, and also for using either of these measurements to predict exposure rates due to the material in buildings. The index corresponds to the terrestrial exposure rate 1 m over an infinite hemisphere of material. The upper limit of the exposure rate in buildings is 2 I, corresponding to a measurement within an infinite sphere of the material. More typical values are 0.8 I in 1-family homes and 1.4 I in multi-story masonry buildings. The coefficient of I depends on the location of the material relative to the measurement point; sample values are given in Figs. 1 and 2. To these values must be added the terrestrial and cosmic ray exposure rates as attenuated by walls, floors and ceilings.

The ^{222}Rn concentration in building air due to construction materials can be computed from the ^{226}Ra concentration in the material, the amount of material, the emanation fraction of ^{222}Rn, the volume of air, and the ventilation rate. Because ^{226}Ra is one of three radionuclides that contribute to the index, the index can only be used to indicate the upper limit of ^{222}Rn concentration in air. If needed, the actual ^{226}Ra concentration can be determined by sampling the material or estimated by performing γ-ray spectral analysis in structures.

Because of the difficulty in determining reliable WL values in building air relative to the ease in measuring γ-ray exposure rates, it is recommended that any extensive surveys that are undertaken measure the latter, and then infer the upper limit of WL values. Only in cases where very high γ-ray exposure rates indicate the possibility of significantly elevated ^{222}Rn concentrations in building air and hence high WL values from materials will it be necessary to perform ^{222}Rn or WL surveys. To utilize this approach, the amount of source material and minimum ventilation rates must be determined for buildings that contain high-^{226}Ra material. It is also suggested that more information be obtained concerning the mechanism of ^{222}Rn exhalation from materials so that the emanation fraction and conditions of maximum exhalation can be known for the materials under consideration.

Acknowledgments—We thank M. D. Matheny, A. Aissi, and C. Hodgson for making many of the field measurements, B. A. Byrne for performing the model calculations, and J. T. Gasper for radioanalytical determinations.

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ANDREWS OFFICE PRODUCTS CAPITOL HEIGHTS, MD (K)

STATE OF ILLINOIS
DEPARTMENT OF NUCLEAR SAFETY
1035 OUTER PARK DRIVE
SPRINGFIELD, IL 62704
(217) 785-9900

THOMAS W. ORTCIGER
DIRECTOR

JIM EDGAR
GOVERNOR

January 17, 1992

Kerr McGee
Mr. Mark Krippel
PO Box 548
West Chicago, Illinois 60185

Dear Mr. Krippel:

This letter transmits the information you requested. Also included in this transmittal is a Status Report on Radon in Illinois. This information is provided in order to supply background information relative to the results.

Please contact me at (217) 786-6398 if you desire further information.

Sincerely,

Robert C. Whitcomb, Chief
Division of Radioecology
Office of Environmental
Safety

RCW:mm
encs.

Radon results for West Chicago, IL

1.8	AVG	5.1
6.5	MAX	13.4
2.2	MIN	1.8
13.4	NUMBER	14
4.4		
2.8		
3.6		
3.9		
9.3		
5.6		
3.1		
3.7		
9.1		
2.1		

Radon results for Ottawa, IL

1.2	AVG	4.4
1.8	MAX	15.5
5.7	MIN	0
3.7	NUMBER	61
2.2		
3.6		
5.6		
0.9		
8.1		
5.2		
3.3		
2.5		
6.6		
1.5		
9.4		
1.2		
4.1		
7.6		
2.1		
3.7		
7.3		
0		
3.9		
5.2		
9.4		
1		
2.2		
2.3		
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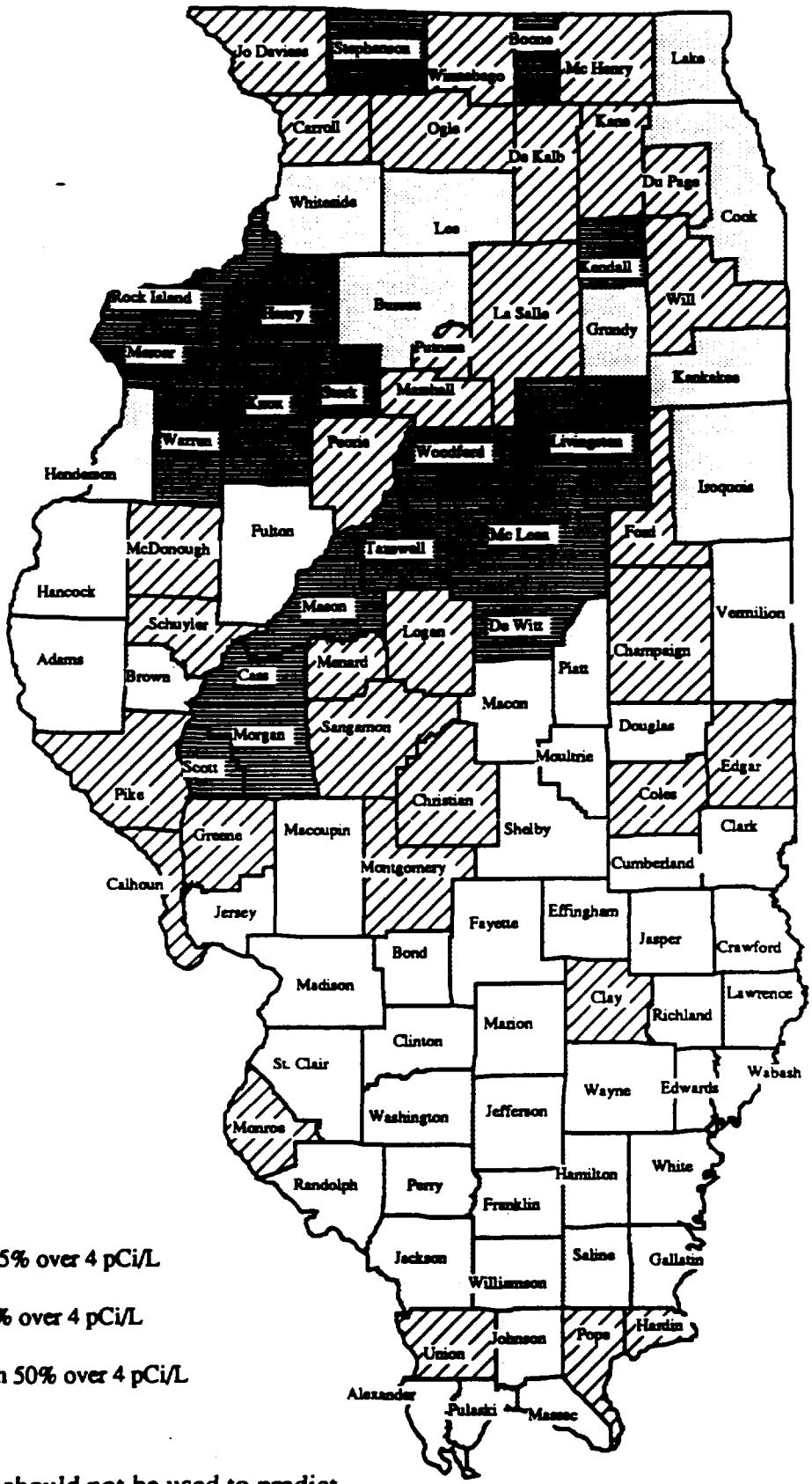
3.7		
4.1		



Illinois
Screening
Program

July
1991

○ Less than 25% over 4 pCi/L

▨ 25% to 50% over 4 pCi/L

■ Greater than 50% over 4 pCi/L

Note: This data should not be used to predict radon concentrations in any particular structure.

ANDREWS OFFICE PRODUCTS CAPITOL HEIGHTS, MD (K)

RADON IN ILLINOIS
A STATUS REPORT

ILLINOIS DEPARTMENT OF NUCLEAR SAFETY
1035 Outer Park Drive
Springfield, Illinois 62704

1990 Update

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SUMMARY

The Illinois Department of Nuclear Safety (IDNS) has performed radon screening measurements in approximately 4,100 homes in 98 counties. Results indicate about 39 percent of the basements tested have radon levels that exceed the U.S. Environmental Protection Agency (EPA) guideline of 4 picocuries per liter (pCi/L) and 1 percent have levels greater than 20 pCi/L. About 11 percent of first floor areas tested have levels greater than 4 pCi/L and less than 1 percent have levels greater than 20 pCi/L. In total, about 31 percent of all homes tested have radon levels greater than 4 pCi/L. If these results represent the entire state, this could mean as many as one million homes in Illinois have levels above EPA guidelines.

The screening program has not indicated any areas in Illinois that face a serious health risk from radon, but there are some areas with a significant percentage of homes with screening results in excess of 4 pCi/L, which merit additional study. Radon may, however, cause significant economic problems for those homeowners with homes greater than the standard. Correlations between house construction characteristics and radon concentrations show no particular feature or combination of features clearly contributes to high radon concentrations.

Although radon concentrations in Illinois are not as high as in some other states (e.g., Pennsylvania), there is still the potential for a health hazard needing to be addressed by IDNS and other agencies. Publicity has increased public concern about radon, proper methods for measuring radon levels and the ability of private companies to provide effective services for reducing levels of radon. There is also considerable concern over the need for and quality of radon measurements conducted when required for real estate transactions. IDNS is assisting the public in coping with these issues. Additional efforts which should be undertaken by IDNS include follow-up studies in neighborhoods identified as potentially exhibiting elevated levels of radon, and development of a training and certification program for radon mitigation contractors.

INTRODUCTION

Because of the significance of radon, Governor James R. Thompson established a task force in June 1986, to investigate the problem of indoor radon in Illinois and report its findings and recommendations. The task force recommended that IDNS be designated the lead agency in the development, implementation and coordination of a comprehensive statewide indoor radon monitoring program. Since the task force recommendations were announced IDNS has conducted studies to (1) locate houses in Illinois with high radon levels; (2) estimate the number of houses in Illinois that might have elevated radon levels; (3) assess the range of indoor radon exposure to Illinois citizens; and (4) determine if any geographic regions that, because of particular geological or other conditions, have greater potential to increase public radon exposure (Illinois Department of Nuclear Safety, 1986). The current Illinois radon program also addresses the question of radon exposure potential in nonresidential structures such as schools and is involved in radon reduction projects and public education programs.

NOTES ON 1990 UPDATE

This report is an update of the November 1988 version of "Radon in Illinois, A Status Report" (Illinois Department of Nuclear Safety, 1988). The Radon Mitigation Act of 1989 requires IDNS to submit a report to the General Assembly describing its findings and recommendations regarding the existence and nature of the risk from radon in dwellings and other buildings in Illinois. This update is intended to serve that purpose. The 1990 report contains new information on:

- Illinois residential screening project;
- epidemiological studies;
- lung cancer risk factors published by EPA;
- the impact of radon measurements on real estate transactions;
- new radon detection equipment;
- Illinois legislation;
- IDNS sponsored training; and
-the State Indoor Radon Grant program

SOURCES OF RADON

Radon is a radioactive gas produced by the decay of radium, a naturally occurring radioactive element. Radium is present as a trace element in all soils and rocks throughout the United States, but is found in high concentrations in some areas. Since it is present in all soils, no areas of Illinois are expected to be entirely radon free. Every Illinois citizen is therefore exposed to indoor radon to some degree. Current studies are designed to evaluate the range and magnitude of radon exposure and contributing factors.

Radon is a colorless, odorless, tasteless and chemically inert gas which produces no apparent symptoms upon immediate exposure. It is denser than air, which means it will tend to sink to the lowest levels in a house in the absence of forced circulation. Because, like helium, it is not chemically active, radon diffuses through porous materials, such as soil, and easily passes through cracks in concrete and other solid materials. Radon-222 has a halflife of 3.8 days, and decays into a series of radioactive elements called radon decay products. These decay products include polonium-218, lead-214, bismuth-214 and polonium-214, and differ from radon in a variety of ways. Radon decay products are solids, and are strongly attracted to surfaces, therefore attaching themselves to walls, floors and dust particles. Since they attach to dust particles, they are removed from the air by particulate filtration. They are also emitters of alpha radiation, and when attached to dust particles, may be inhaled and deposited in the lung, leading to alpha radiation exposure of the lung. Scientists have concluded that long-term alpha exposure of the lung can cause lung cancer, but there is considerable discussion regarding the ability of very low radon levels to cause lung cancer.

UNITS OF MEASUREMENT

Because radon and its decay products have different properties, they have different units of measurements. Radon gas is measured in picocuries per liter (pCi/L). The unit "working level" (WL) is used to express the concentration of radon decay products. Since radon transforms to these decay products, the concentrations of radon and its decay products are equal and the decay products reach their maximum concentration only under equilibrium or "closed box" conditions. These conditions are rare in nature, however. For simple indoor measurements, 50 percent equilibrium is assumed, and it is generally accepted that 200 pCi/L of radon gas is equivalent to 1 WL. Measurements made outdoors, in mines, or under conditions when dust particle concentrations are artificially low, require that working levels be measured directly. A working level monitor is appropriate for this purpose.

A working level month (WLM) is a unit of cumulative exposure first used to describe exposure of uranium miners to radon decay products. One WLM is equivalent to continuous exposure to 1 WL for 170 hours. The current occupational exposure standard set by the Mine Safety and Health Administration is 4 WLM per year. Unless otherwise specified in this report, the term radon is meant to include both radon and its decay products.

FACTORS INFLUENCING RADON LEVELS

Radon levels in houses fundamentally depend on two parameters. These are the rate of gas entry into the house from soil and the rate at which the radon escapes from the house. Among factors contributing to the entry of radon from the soil are the concentrations of radioactive elements in the underlying soil; soil permeability and moisture content; and pressure differentials between the house and the soil. Under conditions where house pressure is less than soil pressure, radon will flow into the house at a higher rate than normal diffusion. Pressure differentials are influenced by weather conditions, heating, ventilation and air conditioning systems, household appliance use and house construction features.

An example of the effect of the barometric pressure alone is shown in figure 1, where the radon level on a lawn was measured by IDNS over a three-day period and found to vary over an order of magnitude. In addition, IDNS measured variations of similar magnitude in indoor radon decay product levels. In this case, a working level monitor was used because this instrument is capable of providing hourly readings. The data for two houses chosen for illustrative purposes only, shown in figures 2 and 3, demonstrate the variability with time as well as the variability from one house to another.

Construction features increasing the rate of radon entry into houses are open sump pumps, basement drains and full or half-size crawl spaces that are open to the underlying soil. Any of these features, when associated with a pressure differential between the house and soil, will tend to increase the rate of radon entry.

Energy efficient homes minimize the exchange of air with the outdoors, thereby conserving heat in the winter and cool air in the summer. These conditions also inhibit radon escaping from the house. In a previous attempt to correlate radon levels with these conditions, however, Nero found energy efficient homes do not necessarily have elevated levels of radon, although a reduction in air exchange rates may increase already elevated levels (Nero, 1984). Low air exchange rates will also tend to distribute the radon more evenly within the house. Current data by Cohen and Gromicko indicate that older, draftier houses probably do not trap and retain radon well and therefore do not usually exhibit elevated radon levels (Cohen and Gromicko, 1988).

The correlation between high radon concentrations and the geology of the underlying soil has been the subject of much investigation since the issue of indoor radon exposure gained national prominence in 1985. This was triggered by the identification of houses with high indoor radon levels in eastern Pennsylvania. These homes were located on a geological structure known as the Reading Prong, which in part consists of formations of uranium-bearing granite. This distinct geological contribution to the radon concentration in this area focused national attention on geology as a major contributing factor to radon exposure.

Despite the number of studies conducted to date, much of the data are inconclusive or contradictory. Of the many factors discussed above, few are well understood. Thus, it is difficult to predict radon levels in any particular home with certainty.

HEALTH EFFECTS OF RADON

Numerous studies of underground uranium miners exposed to radon decay products show an increased risk of lung cancer in comparison with populations not occupationally exposed. The results of epidemiological studies of miners in several countries have been compiled and analyzed by many scientific and regulatory groups.

One of the most recent and comprehensive reports on the health effects of radon and its decay products was published by the Committee on the Biological Effects of Ionizing Radiation of the

National Academy of Sciences (National Academy of Sciences, 1988). The committee reported that although the hazards of exposure during mining are well recognized, the hazards of exposure in other environments such as homes, schools and offices are not yet quantified adequately. Mathematical expression of the risk is complicated by other factors such as cigarette smoking, which is also linked to cancer. The American Cancer Society estimates smoking is responsible for 83 percent of all lung cancers (American Cancer Society, 1987). Many scientific groups are investigating the theory of a synergism between tobacco smoke and radon increasing lung cancer risk.

The National Council on Radiation Protection and Measurements also reviewed health data from miners and developed a model for predicting lung cancer deaths from exposure to radon decay products. This model predicts about 130 fatal lung cancers per million person-WLM, which means the lung cancer risk over a lifetime of exposure to 4 pCi/L of radon is about 1 chance in 119 (National Council on Radiation Protection and Measurements, 1984).

U.S. EPA RISK FACTORS

The EPA is charged with responsibility for setting national standards for exposure to radiation and assumed the federal leadership role for indoor radon issues. The agency recommended a remedial action level of 4 pCi/L for residences. While the EPA believes that no level of radon can be considered risk free, the agency is advising homeowners to reduce the levels of radon in their homes as low as possible, within the limits of the state-of-the-art in radon reduction technology. EPA, using its own methods for calculating cancer risk, estimated that 22,000 of the 130,000 annual lung cancer deaths may be due to exposure to indoor radon. The EPA model predicts about 360 fatal cancers per million person-WLM (U.S. Environmental Protection Agency, 1990). This is somewhat higher than the BEIR IV model estimate of 305 fatal cancers per million person-WLM (National Academy of Sciences, 1988).

In summary, although most scientific groups agree evidence supports the link between radon exposure and an increased risk of lung cancer, there is still considerable discussion over quantitative estimates of the risk, particularly at lower concentrations typical in residences. This is consistent with disagreements among scientists regarding links between low levels of ionizing radiation and other types of cancer.

EPIDEMIOLOGIC STUDIES

There is considerable debate over the preferred method for conducting epidemiologic studies for determining the health risks of radon exposure. EPA has criticized studies for alleged design flaws other than those which show a correlation between radon exposure and lung cancer. In a recent article, Conrath from EPA dismissed the findings of epidemiologic studies that are group studies (Conrath, 1990). An example of such a study was conducted by Cohen, who compared average

radon concentrations with lung cancer death rates by county (Cohen, 1988). Conrath cites the failure of such studies to recognize the mobility of American families and to recognize the wide range of radon results which can be obtained within a given county.

The New Jersey Department of Health recently completed a case-control study of radon in lung cancer among New Jersey women (New Jersey Department of Health, 1989). This study is one of the first large-scale studies based on actual measurements in homes and associated smoking data. The study found a weak correlation between radon exposure and lung cancer for exposed women. Although the authors acknowledge a number of problems with their own data and the data collection approach, Conrath prefers such case-control studies to group studies.

The IDNS screening study data has been targeted by investigators wishing to conduct group epidemiologic studies using Illinois cancer death statistics. IDNS has thus far declined to participate in such studies or supply data for such studies due to the severe limitations of the data set.

DETECTION AND MEASUREMENT OF RADON

There are two kinds of measurements: screening and follow-up monitoring. Screening measurements are taken quickly to determine whether a house has a potential radon problem. These measurements are usually done in the lowest living area of the house, in order to detect the highest level. According to EPA protocols, if the screening result is greater than 4 pCi/L, follow-up measurements are recommended. The objective of follow-up measurements is to estimate the long term average radon concentration in living areas so that informed decisions can be made about risk and the need for remedial action. This is done by placing detectors in at least two locations of high occupancy. For homes with a screening result between 4 and 20 pCi/L, long term (12 month) monitoring is appropriate. For homes with screening results above 20 pCi/L, immediate follow-up measurement (1 to 9 month) is urged.

There are many types of detectors commercially available to measure radon and its decay products. The most common types of detectors used for radon screening are charcoal canisters and alpha track detectors. Charcoal canisters are placed in buildings for a short time period, usually two to seven days, to trap radon gas. The canisters are then sent to a laboratory for analysis of radiation emitted from the trapped radon. This detection method is quite sensitive, but does not allow for monitoring over a long period of time. Thus, individual peaks or valleys in radon concentration like those shown in figure 2 could be measured instead of meaningful average concentrations. Alpha track detectors contain a strip of plastic film inside a canister, and directly measure the radiation emitted by the radon. Since the radiation exposure is cumulative, these can be used for long periods of time, typically from one to 12 months. Alpha track detectors are less sensitive than charcoal, but can be used to obtain average concentrations over longer periods. The exact time they are left in a house is less critical than with charcoal canisters. The long-term average concentrations can be used to estimate radiation exposure levels, whereas the short-term values measured by charcoal canisters cannot be converted to reliable exposure levels of the occupants of a house.

IDNS studied a group of 25 homes with both charcoal and alpha track detectors placed side by side. The frequency distributions for the two types are shown in figure 4. They show similar, although not identical patterns. The shape of the curves highlights the finding that while 43 percent of these homes screened exceeded the EPA guideline, very few had significantly elevated levels. The charcoal curve exhibits a wider range of results, which reflects the inability of charcoal to integrate and average over a long time period. Data for the individual homes are shown in table 1. Note that a charcoal canister significantly overestimated the radon level in house number 17, while the concentration was underestimated in house number 18. Because of this limitation, and other logistical problems associated with charcoal canisters, IDNS performed most of its screening with alpha track detectors.

One of the newest radon detection devices is called the electret ion chamber (EIC). The EIC was described by Kotrappa (Kotrappa, 1988) as using an electrically charged disk in conjunction with a conductive volumetric flask (ion chamber) to indirectly measure the concentration of radon in ambient air. The electret is a true radon integration method and can detect radon at a wide range of concentrations. The electret is treated to hold an electrostatic potential that attracts oppositely charged ions collecting on the electret surface, thereby reducing the electrostatic potential. The decrease in surface potential measured after exposure is directly related to the radon concentration over the sampling period. The ion chamber acts as a collection chamber and is designed to allow radon gas to enter the unit by diffusion.

Electrets were used to pass the EPA proficiency testing program. Results can be read directly in the field or the units can be stored intact and analyzed later. Accuracy of the monitors is not affected by normal temperature or humidity. The same chamber can be used with a different electret for a length of time ranging from two days to one year. The instruments are small, portable, rugged and easy to use. The monitors are reusable down to approximately 200 volts. Although an electret with chamber can be purchased for $60, a reader is $1500. Sensitivity to background gamma radiation requires background checks and subtraction of that voltage due to the gamma component. An interim protocol was established by EPA for the use of EICs (U.S. Environmental Protection Agency, 1989). IDNS recently purchased a set of electrets, and will report on experience with these devices in a future report update.

Follow-up measurements can be done by using alpha track detectors over a 12-month period, or can be done using a series of four quarterly measurements. Quarterly measurements can be taken using a continuous working level monitor (CWLM), or can be done using 4-day charcoal detectors. A CWLM is an electronic device which measures radon decay products, provides results on an hourly basis and integrates over a 4-day period. This instrument samples the ambient air by filtering airborne particles and then counting the alpha particles produced by the radon decay products. It may be important to conduct long term follow-up measurements due to seasonal variations. Winter radon concentrations are typically twice as high as summer concentrations, as illustrated in table 2.

This table also supports the EPA recommendation that follow-up measurements be conducted on all occupied levels of the house, since basement radon levels usually substantially exceed those on the first and second floor.

One advantage of the CWLM is that it can be used for detecting localized radon sources on-site. Another is that it can be used to monitor fluctuations in radon concentration with respect to time. The CWLM does, however, require a trained operator, is expensive and is not easy to calibrate. Most of the IDNS follow-up measurements were done using alpha track detectors, but a CWLM was used in situations where diagnostic or quick measurements were needed.

MEASUREMENTS MADE FOR REAL ESTATE PURPOSES

It is becoming common for residential real estate transactions in the Chicago suburbs to include a radon test. This test is commonly called for in cases where a relocation company is involved, but is being specified by potential private buyers as well. Many of these tests are requested on short notice and must be done using short term devices such as three day charcoal canisters. Some tests are conducted on extremely short notice and are done using grab samples or other 24 hour methods. This precludes any long term measurement and bases the decision on a screening measurement rather than an annual average measurement. In this case 4 pCi/L on a short term screening has become a *de facto* standard and protocol for the sale of a property. Both EPA and IDNS discourage the use of such short measurement results as the sole basis for decision making. This is perhaps the most serious problem facing radon measurement professionals today. A consensus standard and protocol is needed for conducting measurements necessary to satisfy the provisions of a real estate contract.

THE ILLINOIS RADON SCREENING PROGRAM

IDNS designed its radon program as a joint state/local effort wherever possible. To facilitate this effort, training programs for local government personnel were held in areas where these groups were interested, and radon monitoring was conducted as a joint study. IDNS completed such training programs in the city of Chicago and in more than 80 counties throughout the state, usually involving local or regional public health or environmental health agencies.

The first phase of the program was screening Illinois residences using alpha track detectors. The detectors were deployed for no less than two weeks, but no greater than three months. For logistical purposes, the statewide screening was conducted on a county-by-county basis. The number of detectors placed in each county was determined by using geographical and population density considerations but limited by the resources of the department. A minimum of 30 detectors were placed in each county screened with at least one detector per township. In counties with city populations representing a majority of the county, the city was allocated an additional 30 detectors. Greater numbers of detectors were allocated to the six northeastern counties, due to a high population density. The number placed was proportional to the county population.

IDNS SCREENING PROTOCOL

Detectors were placed in basements whenever possible according to EPA protocols. Houses with no basements were screened using first floor measurements. Some of the houses received detectors for both the basement and the first floor so that a basement/first floor ratio could be calculated. This was important in developing an expression for the average annual exposure. Most of the measurements were taken during the heating season. Although homeowners were not instructed to create artificial closed-house conditions, as they would during a 2-day charcoal screening, it is assumed that most homeowners kept their doors and windows closed during the winter.

Homeowners participating in the screening were interviewed using a questionnaire that included questions on the structural features of their homes and use of living areas and appliances. The results of the interviews were compiled and related to the results of the screening measurements. Screening measurement results were forwarded to the homeowners and to IDNS.

EPA recommends follow-up measurements for any house which has a screening result of greater than 4 pCi/L, and a decision to mitigate be made on the basis of an annual average exposure. This value can be measured using a single alpha track placed for a year or can be made using a series of shorter measurements. The higher the exposure rate, the sooner mitigation should be performed. IDNS recommended homeowners conduct follow-up measurements in any home which had a screening result of 4 to 20 pCi/L. For homes which had a screening result greater than 20 pCi/L, follow-up measurements were offered by the department to verify the screening result and to determine whether radon mitigation efforts should be recommended.

To standardize this process, the average annual exposure of the residents was calculated using a weighted average of the winter basement screening result and the spring living area follow-up measurements. The relative weights were based on comparisons of 728 three-month measurements with year-long measurements made in the Reading Prong area (Granlund and Kaufman, 1988). If the weighted average was greater than 8 pCi/L, then the homeowner was advised to take remedial action without further delay. If this average was between 4 pCi/L and 8 pCi/L, then an additional six-month measurement was recommended. Combined results of all measurements were then used to determine whether mitigation was indicated.

SCREENING RESULTS AND DISCUSSION

As of September 1990, IDNS had performed screening measurements in 4,063 homes in 98 Illinois counties, as illustrated in figure 5. These screening data are summarized in table 3 and are broken down by county in table 4. The current data indicate 39 percent of the basements tested have radon levels that exceed the EPA guideline of 4 pCi/L and 11 percent of the first floor areas have such levels. In all, 1,263 measurements taken exceeded 4 pCi/L. This is about 31 percent of the total.

The sample of houses screened to date is a small fraction (about 0.16 percent) of the 2.5 million privately owned houses in Illinois, but if this sample is representative, about 775,000 of the houses

in the state may have elevated basement levels and 275,000 houses may have elevated first floor levels. Since this is a significant number of homes from both a public health and an economic standpoint, and since there are yet no methods that reliably predict the radon concentration in a given house, IDNS continues to recommend that all homeowners conduct radon tests. The frequency distribution of the data is shown in figure 6. The data suggest a log-normal distribution. This is in agreement with the data presented in figure 4, and is consistent with Cohen's analysis of data taken nationwide (Cohen, 1986).

RESULTS OF OTHER STUDIES

The EPA conducted a seven state joint EPA/state radon screening program in 1988 (U.S. Environmental Protection Agency, 1988). The study indicated that from 7 to 45 percent of the houses in those states have the potential for elevated radon levels, as compared to the current Illinois combined estimate of 38 percent. IDNS plans to participate with the EPA in a joint screening during the 1990-91 heating season. The results obtained during the EPA study cannot be compared directly to those obtained by IDNS because the EPA studies are performed using charcoal canisters.

Earlier results compiled by a major supplier of alpha track detectors showed 30 percent of all radon measurements across the country were above the 4 pCi/L level (Terradex, 1988). These results are in good agreement with the radon levels in Illinois homes. The average concentration of indoor radon in this study, 3.9 pCi/L, is approximately equal to the EPA guideline.

EFFECT OF HOUSE CONSTRUCTION CHARACTERISTICS ON INDOOR RADON

A closer examination of the distribution of radon results by house construction characteristics was done to develop a better understanding of the behavior of radon in various types of homes. The following information was provided by homeowners and compiled in a database along with the screening results:

- age of house;
- type of substructure (basement, slab or crawlspace);
- primary heating source (gas, oil, electric, others);
- basement characteristics such as cracks or drains; and
- crawlspace characteristics such as exposed earth.

Homeowners were also asked to rate their home subjectively according to its energy efficiency on an arbitrary linear scale.

An attempt was made to correlate these features and characteristics with either high or low radon concentrations. Results are presented in table 6. There are weak correlations between some of the features studied and radon levels and absolutely no correlation with other features which are commonly thought to influence radon levels.

The age of the house was not a good indicator. Homes less than 15 years old should be more energy efficient than older homes but no increase in radon concentration was found in these homes. On the other hand, homes greater than 50 years old are thought to be drafty but were not significantly lower in radon concentration. Unfortunately more than 86 percent of the homeowners in the study rated their home energy efficiency as "good" or "excellent;" so little could be drawn from this information, although the average level in these houses (4.0 pCi/L) was slightly higher than those rated "not at all" or "somewhat" energy efficient (3.1 pCi/L).

Although successful radon mitigation efforts almost always depend on a well-sealed basement floor, there was little evidence that houses with basement floor leaks and cracks automatically have high radon concentrations. The presence of exposed earth either in a basement or accessible crawlspace seemed to be a common factor in many of the higher concentration homes. Homes with crawlspaces that are fully ventilated and not accessible from the basement tended to be lower in radon than the average.

Several studies have failed to show a correlation between certain home construction features and high radon concentrations. A survey conducted by Cohen of 453 houses in 42 states found only weak correlations between radon levels and home construction features (Cohen, 1986). One of Cohen's conclusions was that geological factors might control radon levels to a greater degree than construction features. This poor correlation precluded public health officials from focusing efforts on specific types of houses or ruling out radon problems for significant numbers of homeowners.

EFFECT OF GEOLOGICAL FACTORS ON INDOOR RADON

It is not clear whether there are any particular geological formations in Illinois which contribute to high radon exposures. There is no evidence of any areas with radium concentrations similar to those in the Reading Prong area, but radium levels do vary across the state and Illinois soils do exhibit varying permeability and moisture content. Some investigators tried to link the National Uranium Resource Evaluation (NURE) data with indoor radon levels, but the NURE data is useful only for locating uranium and other nonspecific gamma ray anomalies.

Since IDNS did not have the resources to study geological factors directly on a statewide basis, the original approach was to rely on the statewide screening program to identify clusters of homes with elevated radon levels. This was to be done by screening neighborhoods around homes with confirmed radon levels above 20 pCi/L. It was then planned to study the geology in these local areas. Due to lack of resources, this neighborhood screening program was postponed. As indicated in table 3, the department identified about 44 neighborhoods that should be studied.

There are no known areas of the state which exhibit consistently elevated radon levels, such as those found in Pennsylvania. The highest result recorded was 75.6 pCi/L in DeWitt County. Although no other homes in that county were above 20 pCi/L, the average result for the county was about

7 pCi/L. Other very high values were found in the state but they were due to the disposal of radium wastes and not due to natural conditions.

Illinois screening data identified regions of the state that exhibit higher than average radon concentrations. These regions are in north central and northwestern Illinois. IDNS identified 18 counties where the majority of the screening measurements were greater than 4 pCi/L (see figure 7). The Chicago area was not identified as a problem area relative to the rest of the state, but there may be small local areas of higher than average radon. IDNS has attempted to develop a simple description of the geographical boundary of the area of greatest concern. This proved difficult. Note, however, that the area with zip codes beginning with "61" are about twice as likely to have a screening measurement in excess of 4 pCi/L than areas with zip codes beginning with "60" and "62".

RADON IN SCHOOLS

Not all personal radon exposure can be attributed to private residences. Studies are in progress to determine what fraction of personal radon exposure is due to exposure at home. Some factors that allow radon to enter houses also apply to commercial and public buildings. Some public buildings are of particular concern due to potential radon exposure to children. Because of this concern, IDNS initiated a screening program for schools. The program has had two phases thus far. In the first phase, for each of 21 counties screened, two elementary schools were selected for participation. Six detectors were placed in each school with at least two detectors placed on each level. Detectors were placed only in areas frequented by students, such as classrooms, libraries and lunchrooms. Some basement areas fell into this category. Detectors were left in place between one and two months. Screenings, conducted on this limited basis, indicate about 25 percent of the student areas contained radon levels exceeding 4 pCi/L.

Most recently, IDNS performed long term alpha track measurements in all public schools in Clark and Wayne counties. A total of 25 schools were tested. Only one student area had radon levels in excess of 4 pCi/L. Data for all schools are listed in table 7.

IDNS has been involved in screening, follow-up and diagnostic measurements at a group of Peoria schools since February 1989. At that time, IDNS placed 125 EPA charcoal detectors in six schools for a three-day test. The results ranged from 0.5 to 19.6 pCi/L. Follow-up tests were conducted by IDNS using alpha track detectors in 26 student areas that had screening results in excess of 4 pCi/L.

In November 1989 the EPA Office of Research and Development (ORD) proposed a project to perform diagnostic measurements in schools to develop effective mitigation strategies. EPA Region V suggested a group of Peoria schools that were tested during the February 1989 study be considered for the ORD School Diagnostics and Mitigation Strategy Project. IDNS contacted Peoria School District 150 administration, who agreed to participate. EPA and IDNS representatives conducted a walk-through audit and made radon diagnostic measurements at the Harrison, Tyng and Calvin Coolidge schools and determined these schools were suitable for the ORD project.

In February 1990, the IDNS officially proposed to ORD that the Peoria schools should be considered for the project. IDNS staff recommended the radon levels in one room of Harrison and Tyng and three rooms in Calvin Coolidge be reduced to below 4 pCi/l based upon their three-season averages. In May 1990, the ORD team performed the diagnostic measurements in Harrison, Tyng and Calvin Coolidge schools. The team reviewed the diagnostic data and developed a report that recommends an optimum radon mitigation strategy for each school. The report suggests the radon problems are caused to some degree by inoperable HVAC systems.

Schools are not yet required by either federal or state law to test for radon. However, IDNS encourages all schools to conduct screenings for the same reasons home testing is recommended. Some school districts voluntarily tested for radon, but many others are reluctant to do so for two reasons. First, while radon screening costs may be relatively low, school officials do not believe they have sufficient resources to mitigate radon problems if they are discovered. Secondly, since there are no mandatory protocols for radon testing, school officials are concerned that tests conducted now may not be valid once mandatory protocols are adopted. Even when voluntary tests are conducted, school officials are reluctant to disclose results to IDNS. As a result, IDNS has little information regarding the scope and results of voluntary testing.

RADON IN PUBLIC BUILDINGS AND IN THE WORKPLACE

Very little testing in public buildings and workplaces has been conducted. As with private residences, commercial properties are being tested for radon when sold, but there is not a significant effort on the part of employers to characterize employee workplaces. To our knowledge, the Occupational Safety and Health Administration has not made radon exposure a high priority compliance item. More research is needed to determine the nature and extent of radon problems in commercial and industrial structures.

The Illinois Secretary of State (SOS) is the custodian of many of the state government buildings in Springfield. IDNS and SOS conducted a screening study of 26 buildings in Springfield in 1989. The results ranged from 0.3 to 15.2 pCi/L. As a result of this screening, IDNS recommended follow-up measurements be made at three locations. SOS took follow-up steps at all three locations. The most interesting mitigation was conducted in the basement of the state capitol. Grab samples in the electrical shop of the capitol ranged from 13.4 to 21.7 pCi/L. The capitol is a complex structure with underground passageways and ventilation plenums exposed to soil. Very little fresh air was being routed to the shop area. In this case, changes in the HVAC system were needed to solve the radon problem in the shop and bring radon concentration down below 4 pCi/L.

REDUCING RADON EXPOSURE

The objective of the statewide radon program is not only to identify any problems related to radon exposure, but to provide recommendations for remedial action to reduce radon exposure. Most IDNS follow-up studies in houses with elevated radon levels involve evaluating causes, as well as confirming screening measurements. Radon is not only a significant public health issue, but also an

economic issue. If 31 percent of Illinois residences ultimately prove to have levels greater than 4 pCi/L, this translates to about one million homes. The cost of reducing radon levels could range from $200 to $2,000 or more per home, meaning a potential cost of $200 million to $2 billion to Illinois citizens. These cost estimates apply only to private residences and do not include public or commercial buildings.

The EPA prepared guidelines for reducing radon levels in private residences (U.S. Environmental Protection Agency, 1987) but many of these techniques are still in the experimental stage. The selection of an appropriate and cost-effective radon reduction technique requires that the radon entry routes be well understood. This understanding is accomplished by conducting a visual inspection which may be accompanied by local diagnostic radon measurements. Measurements usually indicate whether a simple remediation method will be sufficient or whether several methods need to be combined to achieve acceptable results.

RADON REDUCTION METHODS

Several radon reduction methods have been used successfully in the United States. One simple method to prevent radon from seeping into a house is to seal all cracks and openings in basement floors and walls. Sealing such cracks is often an important preliminary step even when other, more complex methods are used. Sealing cracks alone may result in only moderate to small reductions in radon concentrations. The cost of sealing a basement is typically about $500.

Another simple method is to ventilate a basement or crawlspace to replace the radon with fresh air, which contains radon at lower concentrations. However, an architectural society expressed concern about the applicability of this technique in the colder midwestern climate since natural ventilation could cause frozen pipes during the winter. To conserve heat, a ventilation system can be coupled to an air-to-air heat exchanger to use the heat in the outgoing air to warm the incoming air. Heat exchange units cost about $1,500 including installation.

In cases where ventilation is impractical or not cost effective, it may be necessary to ventilate below the slab or ventilate the block wall of a basement area. This method exhausts radon to the outdoors before it reaches the living space and also helps reduce the pressure differential between the subslab and the basement. All basement cracks must be well sealed in order for this method to be effective. A complete system can cost $2,500.

A similar method for exhausting radon to the outdoors, before it reaches the living space, involves ventilation of drain tiles. Drain tile systems are typically installed to keep water from pooling around the foundation of a home, but can also be used to remove radon gas from this area. When properly used, these systems can be very effective. Installation costs in existing homes can cost $1,500.

While it may seem that some of these devices could be installed by the average handyman, IDNS recommends that homeowners consult with experienced professionals before attempting any radon mitigation.

IDNS EXPERIENCE IN RADON MITIGATION EFFORTS

In 1988 IDNS staff completed a remediation project at a home in Schaumburg. At the request of the village of Schaumburg, IDNS provided technical assistance including evaluation of the radon levels; diagnosis of the source; and routes of entry and recommendations on a reduction method. Grab sample measurements indicated that a basement sump and the heating ductwork beneath the slab-on-grade portion of the house which penetrated the adjacent basement wall were the major entry routes. Sealing the sump hole and other minor radon entry routes was not effective in reducing the basement radon levels to below 4 pCi/L. A drain tile ventilation system using the existing drain tile loop and sump hole was then installed. This active system reduced the radon levels to about 2 pCi/L. Details of this mitigation effort are reported elsewhere (Hamel, 1988).

At the request of the Illinois Department of Energy and Natural Resources (ENR), IDNS monitored radon levels and assisted in a remedial action project at the Springfield Energy House. This house was designed and built by ENR to demonstrate the value of energy efficient building techniques and features. The features include a super-insulated shell to reduce heat loss and an underground ice storage cooling system to provide air conditioning in the summer (Illinois Department of Energy and Natural Resources, 1987). Since it is suspected that homes with low air exchange rates have high radon levels, the house was screened and found to have high concentrations in localized areas. The main route of entry for radon was the penetration from the basement to the ice storage unit. Once this penetration was sealed, an annual follow-up measurement was made. The average general living area concentration was found to be 3.8 pCi/L.

IDNS is concerned about the availability and reliability of radon mitigation contractors. Currently there is no requirement for radon mitigation contractors to register with the state, nor is there a mandatory certification program run by the federal government. IDNS recommends that homeowners employ contractors who have successfully completed the EPA Radon Contractor Proficiency Program. This program is available to Illinois contractors through the Midwest Universities Radon Consortium (MURC). Some radon mitigation work is currently being done by contractors with previous experience in home renovation and remodeling, but whose education and experience in radon detection and mitigation techniques are not known.

PUBLIC EDUCATION PROGRAMS

A major objective of the Illinois program has been to inform and educate the public about radon. As part of this program, IDNS provides basic information about indoor radon and its associated health risks, together with information about radon monitoring. A total of 30 presentations were given between January 1989, and July 1990, on general radon awareness. Another 30 presentations were given in conjunction with the statewide residential radon screening study. These presentations were designed to train local volunteers to place radon detectors in accordance with IDNS protocols and to complete the documentation needed for the study. Because the results of the statewide monitoring program cannot be used to predict radon levels in specific houses, IDNS encourages occupants to monitor their own houses and to report high results to IDNS.

In order to facilitate this process, IDNS distributes a list of firms supplying devices that passed the EPA radon monitoring proficiency test. A variety of additional radon-related instructional materials have been distributed to the public, including over 15,000 copies of the "Citizens Guide to Radon" (1986 edition) prepared by the EPA and reprinted by IDNS.

Information available about radon mitigation is not as valuable. There is a lack of qualified radon mitigation contractors and a lack of methods for evaluating which contractors are qualified. At the federal level, the EPA has started a radon contractor proficiency program, but participation is voluntary and therefore limited. There is no contractor training or certification program in Illinois, nor any requirement for follow-up measurements. IDNS has received citizen complaints against contractors, but the department does not have any regulatory authority over radon mitigation contractors. Both specific regulatory authority and the resources to provide training to contractors would provide significant consumer protection and increase public confidence in the program.

From July 1986, to February 1988, the department funded and staffed a toll-free radon information "hotline" to provide information on radon to Illinois citizens. During this period, an average of 500 calls per month were received. Funding and staffing were suspended for this program in 1988 but resumed in August 1990.

In March 1987, the department sponsored a conference on radon, radium and environmental radioactivity. One full day was devoted to talks on radon in homes, radon risk evaluation, geological considerations, monitoring procedures and mitigation techniques. The conference was designed for Illinois citizens, public health agencies and environmental groups, and was attended by about 500 people.

County and other local government agencies have expressed interest in assisting with public education, but have limited resources to conduct large scale programs. IDNS supplies these agencies with speakers, technical advice and printed information for distribution by their offices.

ILLINOIS LEGISLATION

Two key pieces of radon-related legislation were passed during 1989. The Radon Mitigation Act authorizes the IDNS to establish and coordinate a comprehensive program for detecting and reducing the amount of radon in homes and other buildings in Illinois. The act exempts radon results obtained by IDNS from disclosure requirements of the Freedom of Information Act. This is an important step forward allowing IDNS staff to continue radon studies while protecting the participants' property values. The bill also enabled IDNS to secure independent general revenue funding from the Illinois General Assembly for radon related projects.

House Bill 1611, "An Act in Relation to Radon Testing", authorizes IDNS to establish a registration program for persons selling any device or performing any service for compensation to detect radon or its decay products. The program is intended to regulate those who place passive detectors in

structures or who perform measurements using working level monitors, grab samplers and other active methods. Rules for implementation of this program (32 Illinois Administrative Code 420) were published in the Illinois Register on November 30, 1990. IDNS estimates there will be 300 registrants in this program.

IDNS SPONSORED TRAINING

In anticipation of the implementation of these rules, IDNS and the MURC co-sponsored three training sessions on radon measurements for potential registrants. The sessions were held in Mt. Vernon, Bloomington and Des Plaines during the week of April 9, 1990. A total of 110 people attended, but the sessions were overbooked by a considerable margin. IDNS plans to repeat the sessions as soon as the rules are final.

EPA GRANT

On May 1, 1990, IDNS was awarded a grant under the State Indoor Radon Grants program administered by the EPA. Under the provisions of the grant, IDNS will undertake a greater number of projects than it would using only state funding. Some of these projects include participating in the EPA/state screening program; providing a limited number of free radon detectors to low income school districts identified by the state Board of Education; coordinating a school mitigation demonstration project; and conducting a study of Illinois building codes as they relate to radon resistant new construction.

CONCLUSIONS

1. IDNS has performed radon screening measurements in approximately 4,100 homes in 98 counties. Results indicate about 31 percent of all homes tested have radon levels greater than the EPA standard of 4 picocuries per liter. The screening program identified certain areas in Illinois with significant percentages of homes with screening results in excess of the standard that merit additional study.

2. Schools are not yet required to conduct radon testing. IDNS has little information regarding the scope and results of voluntary testing, but is concerned that the uncertainties regarding costs of mitigation and testing are forcing school officials to postpone testing until it is mandatory.

3. IDNS is providing a wide variety of educational information in response to public inquiries. This effort is, for the most part, a reactive effort and therefore limited in scope. Although radon has received considerable publicity, most members of the public still need basic information about radon. News reports and public service announcements provided by the media have been either misleading or ineffective.

4. The registration and training of persons performing radon measurement services are good initial steps toward assuring consumer confidence in radon services in Illinois. Radon mitigation services are still not covered under the program.

5. Radon reduction in homes is still primarily a post-construction activity in Illinois. There is no significant effort on the part of builders or architects to incorporate radon resistant features in new construction.

6. Radon measurements made for the purpose of satisfying provisions of a real estate contract are not being conducted according to any specific protocols or quality assurance guidelines. This causes considerable difficulty for homeowners whose transactions depend on accurate results. Erroneous results may cause delays in the transaction, or may force a homeowner to install costly mitigation equipment where it is not needed.

RECOMMENDATIONS

1. Complete the radon screening of all Illinois counties. Four counties remain to be screened before the project is considered complete.

2. Conduct follow-up studies in neighborhoods where local clusters of homes with potential radon problems are suspected. This would help to identify localized areas where the geological conditions could be studied.

3. Encourage and support voluntary testing by schools. This could be done by conducting briefings for school administrators, conducting mitigation demonstration projects and by providing free detectors to a limited number of low-income school districts.

4. Continue to develop more active approaches to public education. This might include providing radon information to large numbers of schools and libraries. More effort is needed to educate the media as well. IDNS staff should continue to respond by sending radon information to members of the media and by making department representatives available for interviews.

5. Develop and implement a certification program for persons or companies who perform radon mitigation services. Although EPA conducts a voluntary program, Illinois has no mechanism for formally recognizing participation in the program. In conjunction, IDNS should continue to develop and conduct training programs for those who offer mitigation services as well as measurement services.

6. Evaluate the need for changes in building codes in Illinois, since the construction of radon resistant structures is the only long term solution to the indoor radon problem. Illinois should follow the lead of states in the eastern U.S. that have adopted radon resistant features in building codes.

7. Work with EPA and with the Illinois Association of Realtors to arrive at a consensus regarding protocols and quality assurance associated with radon measurements made for real estate transactions.

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Figure 1

OUTDOOR RADON LEVELS VS. TIME

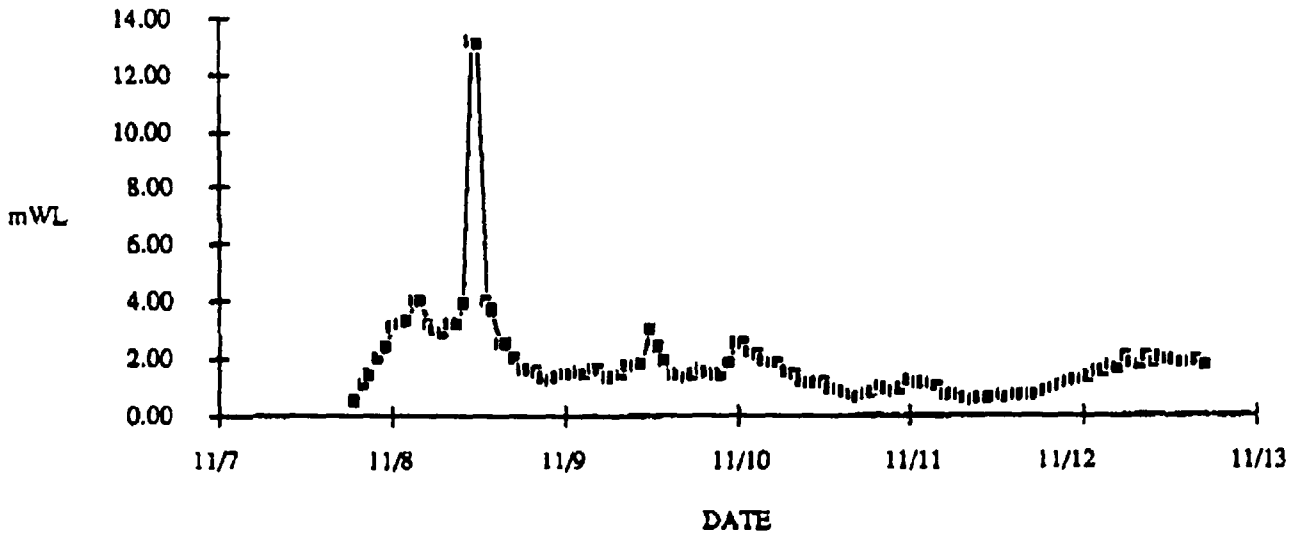


Figure 2

RADON DECAY PRODUCTS VS. TIME IN THE BASEMENT
OF AN ILLINOIS HOME

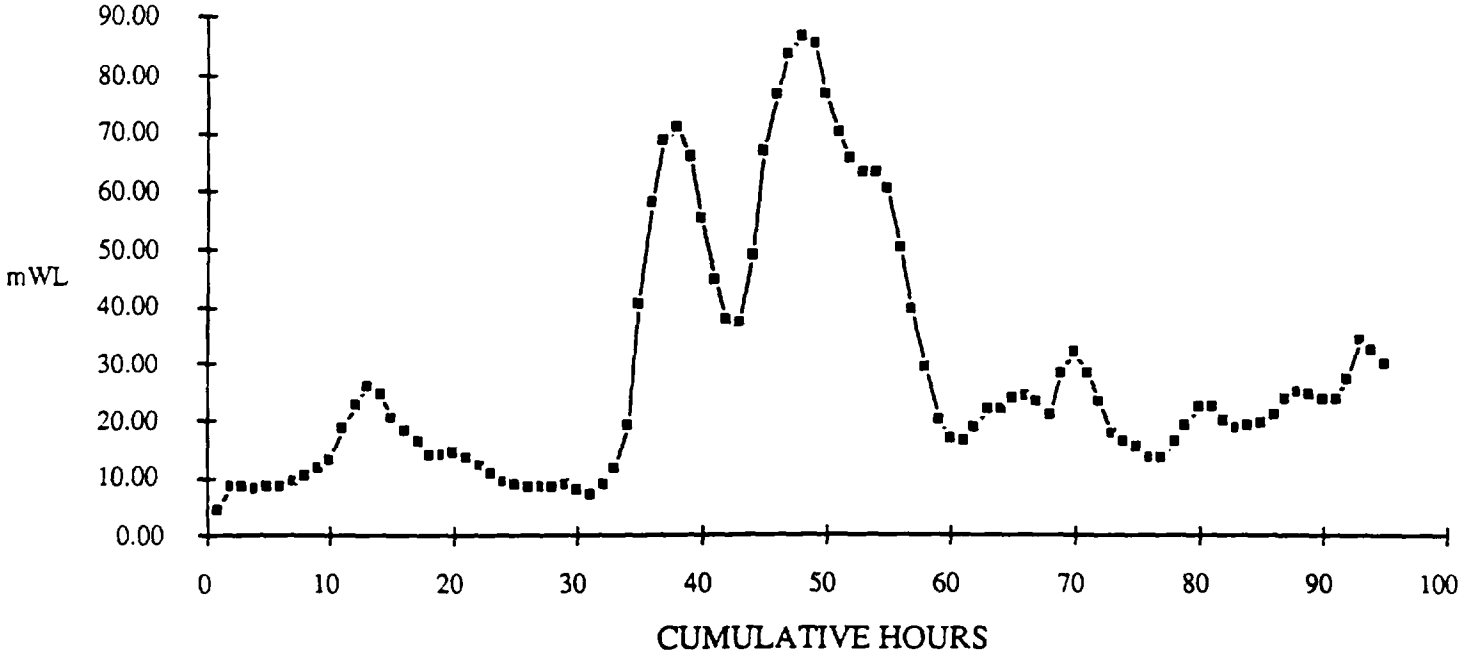


Figure 3

RADON DECAY PRODUCTS VS. TIME IN THE BASEMENT
OF AN ILLINOIS HOME

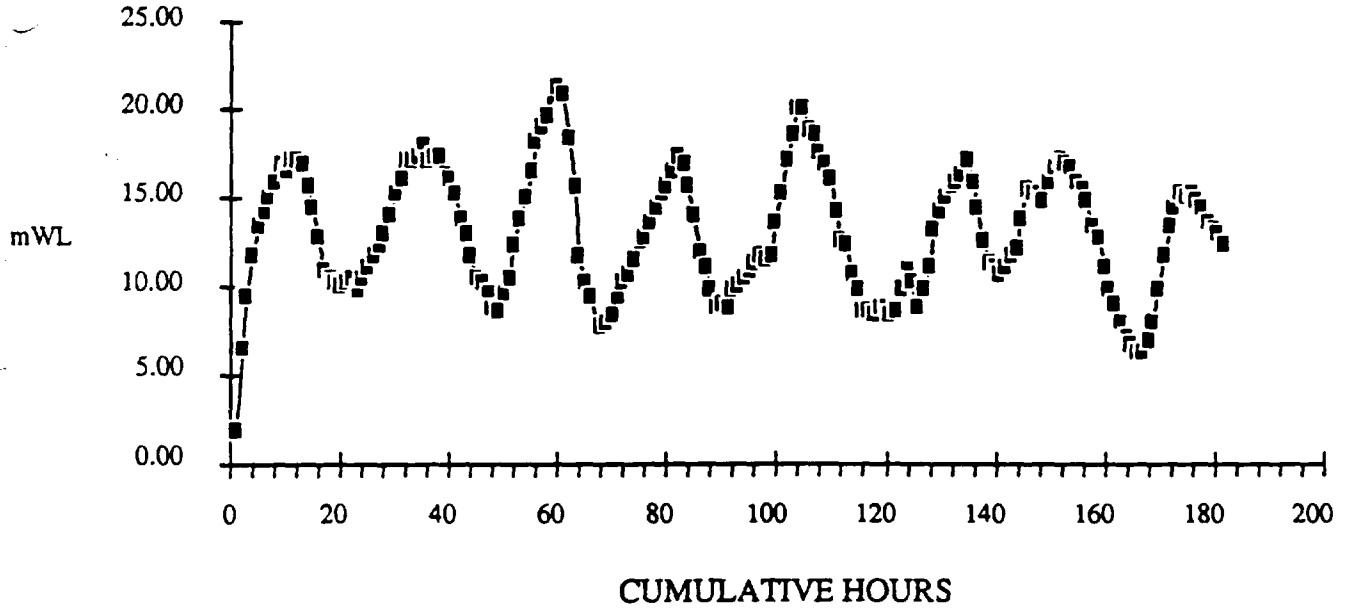


Figure 4

SUMMER BASEMENT RESULTS

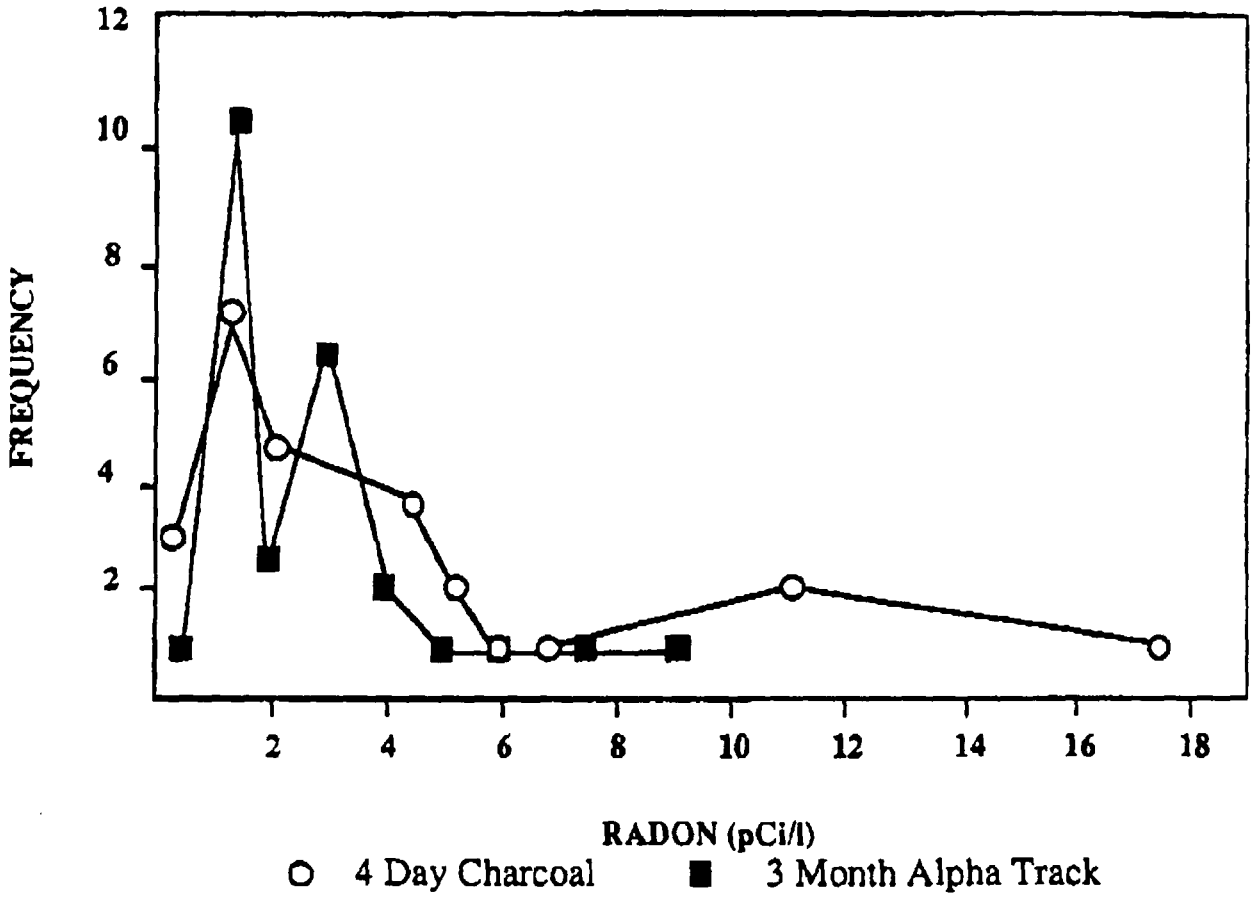


Figure 5

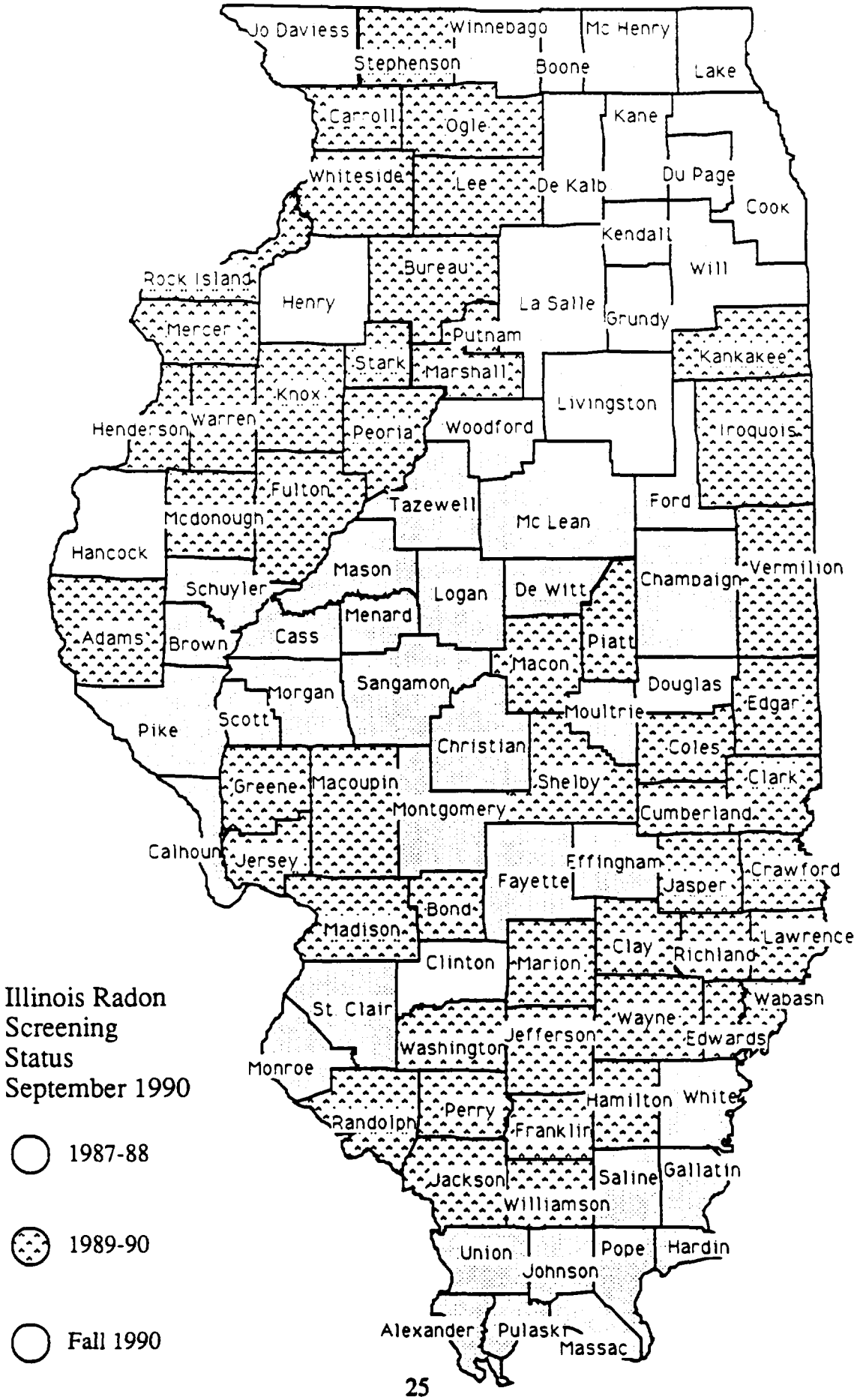

Jo Daviess
Winnebago
Mc Henry
Stephenson
Boone
Lake
Carroll
Ogle
Kane
Whiteside
Lee
De Kalb
Du Page
Cook
Kendall
Will
Rock Island
Henry
Bureau
La Salle
Grundy
Mercer
Putnam
Stark
Marshall
Kankakee
Knox
Livingston
Henderson
Warren
Peoria
Woodford
Iroquois
Fulton
Tazewell
Mc Lean
Ford
Mcdonough
Hancock
Mason
Champaign
Vermilion
Schuyler
Logan
De Witt
Menard
Adams
Brown
Cass
Piatt
Macon
Morgan
Sangamon
Douglas
Edgar
Pike
Scott
Moultrie
Christian
Coles
Greene
Macoupin
Shelby
Clark
Montgomery
Cumberland
Calhoun
Jersey
Effingham
Fayette
Jasper
Crawford
Bond
Madison
Clay
Richland
Lawrence
Clinton
Marion
St. Clair
Wayne
Wabash
Jefferson
Edwards
Washington
Monroe
Hamilton
White
Perry
Randolph
Franklin
Jackson
Saline
Gallatin
Williamson
Union
Pope
Hardin
Johnson
Alexander
Pulaski
Massac
Illinois Radon
Screening
Status
September 1990
1987-88
1989-90
Fall 1990

Figure 6

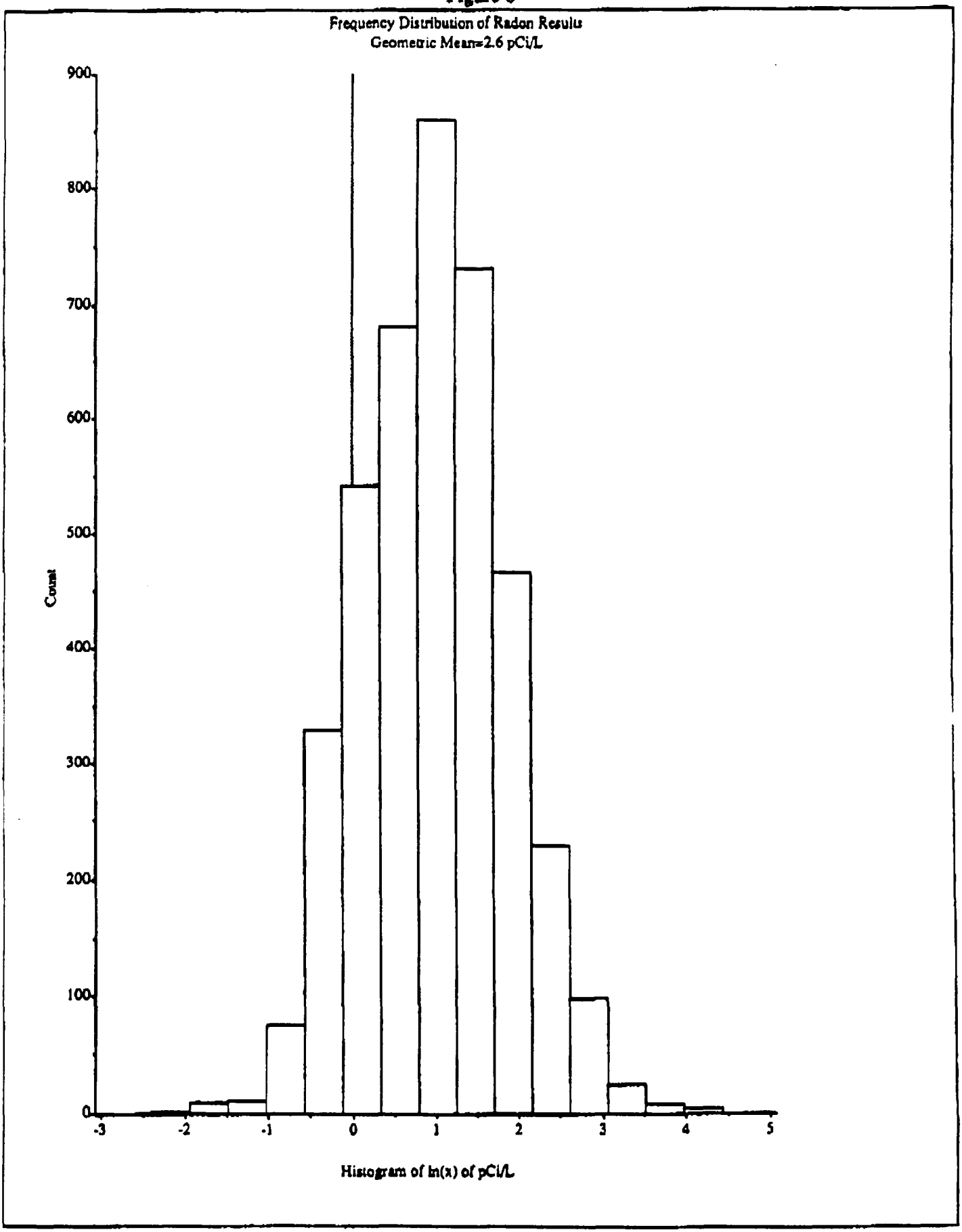
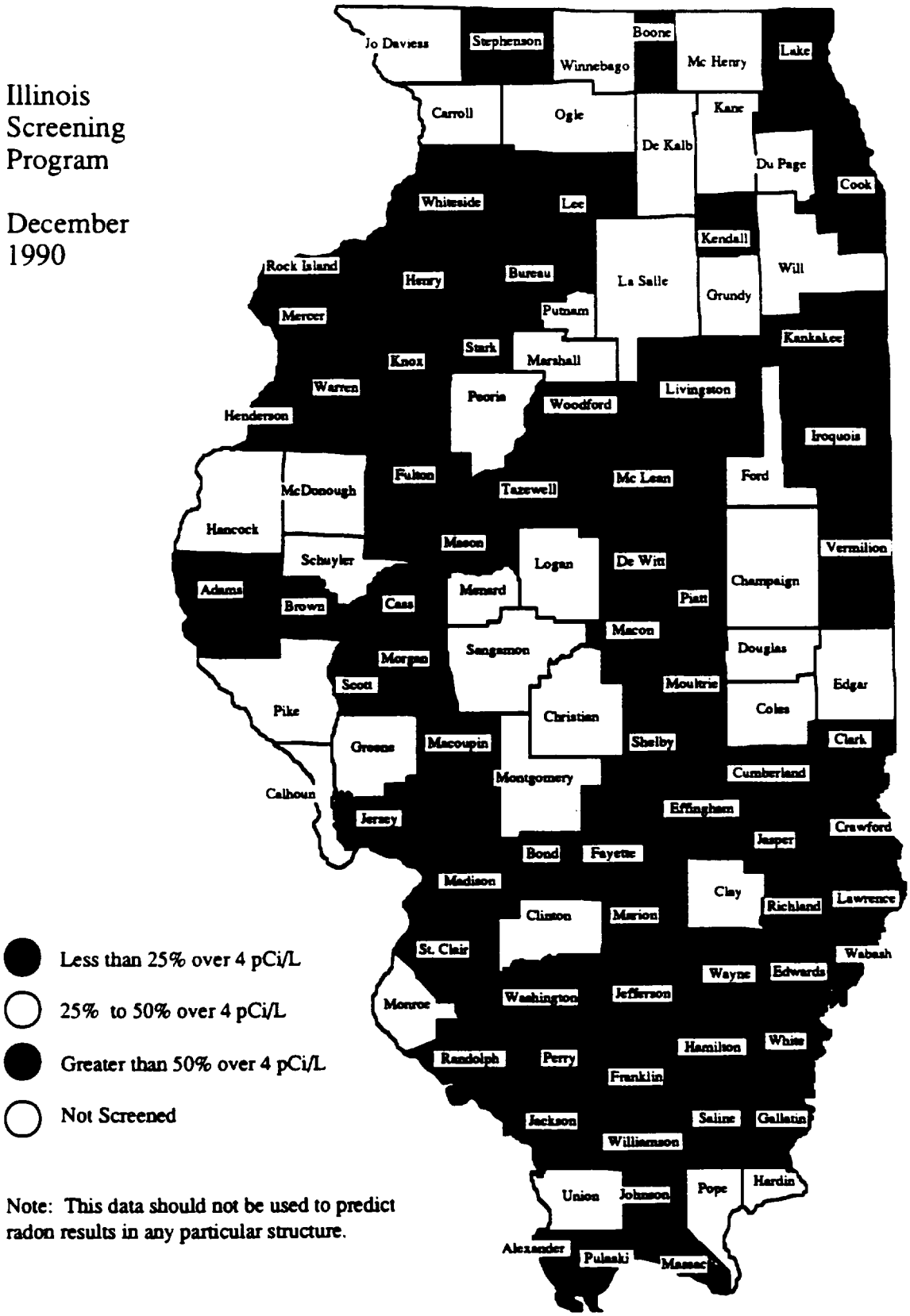

Frequency Distribution of Radon Results
Geometric Mean=2.6 pCi/L
Count
900
800
700
600
500
400
300
200
100
0
-3
-2
-1
0
1
2
3
4
5
Histogram of ln(x) of pCi/L

Figure 7

Illinois
Screening
Program

December
1990



Less than 25% over 4 pCi/L

25% to 50% over 4 pCi/L

Greater than 50% over 4 pCi/L

Not Screened

Note: This data should not be used to predict radon results in any particular structure.

Table 1

SUMMER BASEMENT RESULTS
RADON CONCENTRATIONS IN pCi/L

House Number	Charcoal 4 Days	Alpha Track 3 Months
1	2.4	3.1
2	7.2	4.9
3	1.0	2.4
4	1.3	1.5
5	1.0	1.3
6	4.1	1.9
7	1.5	1.0
8	10.8	8.3
9	2.1	1.4
10	4.0	3.5
11	3.7	3.4
12	2.5	3.2
13	1.0	1.0
14	3.0	1.5
15	1.6	1.4
16	2.0	1.2
17	15.3	7.7
18	0.9	3.8
19	3.8	5.9
20	0.6	1.1
21	1.2	2.3
22	4.7	4.4
23	5.8	4.2
24	1.8	3.1
25	1.0	0.8

Table 2

SUMMARY OF ILLINOIS RADON SCREENING RESULTS BY SEASON

Fall

	#	Min	Avg	Max	#>4	%>4	#>20	%>20
Basement	130	0.7	5.2	28.1	55	42	4	3
First Floor Bedroom	36	1.2	2.2	3.4	0	0	0	0
First Floor Living Area	55	0.5	2.2	5.5	4	7	0	0
Total	221	0.5	4.0	28.1	59	27	4	2

Spring

	#	Min	Avg	Max	#>4	%>4	#>20	%>20
Basement	995	0.3	4.1	55.1	339	34	12	1
First Floor Bedroom	213	0.3	2.1	19.3	19	9	0	0
First Floor Living Area	171	0.2	1.6	13.7	11	6	0	0
Other	24	0.6	2.3	12.2	3	12	0	0
Total	1403	0.2	3.4	55.1	372	27	12	1

Summer

	#	Min	Avg	Max	#>4	%>4	#>20	%>20
Basement	136	0.8	5.2	22.4	68	50	1	1
First Floor Bedroom	49	0.7	1.9	3.9	0	0	0	0
First Floor Living Area	16	0.8	3.6	23.2	3	19	1	6
Total	201	0.7	4.3	23.2	71	35	2	1

Winter

	#	Min	Avg	Max	#>4	%>4	#>20	%>20
Basement	1659	0.1	4.7	75.6	670	40	26	2
First Floor Bedroom	352	0.3	2.6	12.5	62	18	0	0
First Floor Living Area	225	0.09	2.4	18	29	13	0	0
Other	2	3	3.0	3	0	0	0	0
Total	2238	0.09	4.2	75.6	761	34	26	1

Table 3

SUMMARY OF ILLINOIS RADON SCREENING RESULTS BY LIVING AREA

Living Area	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	2920	0.1	4.6	75.6	1132	39	43	1
First Floor Bedroom	650	0.3	2.3	19.3	81	12	0	0
First Floor Living Area	467	0.09	2.1	23.2	47	10	1	0
Other	26	0.6	2.3	12.2	3	12	0	0
Total	4063	0.09	3.9	75.6	1263	31	44	1

Table 4

SUMMARY OF ILLINOIS SCREENING RESULTS BY COUNTY

Adams County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	24	0.5	4.1	10.9	11	46	0	0
First Floor Bedroom	19	0.5	2.4	13.6	2	11	0	0
First Floor Living Area	20	0.5	1.9	10.1	1	5	0	0
Other	2	0.6	0.6	0.6	0	0	0	0
Total	65	0.5	2.8	13.6	14	22	0	0

Alexander County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	17	1.4	3.5	5.6	6	35	0	0
First Floor Bedroom	5	1.6	2.1	2.9	0	0	0	0
First Floor Living Area	3	0.5	1.6	3.3	0	0	0	0
Total	25	0.5	3.0	5.6	6	24	0	0

Bond County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	13	0.5	2.4	6.2	3	23	0	0
First Floor Bedroom	10	0.4	1.7	5.9	1	10	0	0
First Floor Living Area	4	0.6	2.6	5.8	1	25	0	0
Total	27	0.4	2.2	6.2	5	19	0	0

Boone County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	51	1.2	5.1	13.3	31	61	0	0
First Floor Bedroom	1	3.4	3.4	3.4	0	0	0	0
First Floor Living Area	2	3.6	4.2	4.8	1	50	0	0
Total	54	1.2	5.0	13.3	32	59	0	0

Table 4 (cont'd)

Brown County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	19	0.6	5.8	55.1	6	32	1	5
First Floor Bedroom	9	0.6	1.5	4.4	1	11	0	0
First Floor Living Area	1	0.8	0.8	0.8	0	0	0	0
Other	6	0.8	1.9	3.5	0	0	0	0
Total	35	0.6	3.9	55.1	7	20	1	3

Bureau County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	1	3.1	7	5	23	0	0
First Floor Bedroom	1	0.5	0.5	0.5	0	0	0	0
Total	23	0.5	3.0	7	5	22	0	0

Calhoun County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	1.3	3.7	10.8	6	27	0	0
First Floor Bedroom	4	1.3	3.9	5.5	2	50	0	0
First Floor Living Area	3	1.3	1.8	2.9	0	0	0	0
Total	29	1.3	3.6	10.8	8	28	0	0

Carroll County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	25	1	5.5	25.4	12	48	1	4
First Floor Living Area	2	0.7	1.1	1.5	0	0	0	0
Other	1	3.7	3.7	3.7	0	0	0	0
Total	28	0.7	5.1	25.4	12	43	1	4

Cass County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	24	1.8	5.6	11.3	18	75	0	0
First Floor Bedroom	1	1.4	1.4	1.4	0	0	0	0
First Floor Living Area	3	2.2	3.0	4	0	0	0	0
Total	28	1.4	5.2	11.3	18	64	0	0

Table 4 (cont'd)

Champaign County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	16	1.6	5.9	17.6	8	50	0	0
First Floor Bedroom	17	1.2	4.1	8	8	47	0	0
Total	33	1.2	5.0	17.6	16	48	0	0

Christian County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	32	1.8	5.8	15.5	21	66	0	0
First Floor Bedroom	17	0.6	2.4	6.1	1	6	0	0
First Floor Living Area	1	8.7	8.7	8.7	1	100	0	0
Total	50	0.6	4.7	15.5	23	46	0	0

Clark County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	13	0.8	3.3	8.4	4	31	0	0
First Floor Bedroom	9	0.6	1.0	3.3	0	0	0	0
First Floor Living Area	3	0.5	0.6	0.8	0	0	0	0
Total	25	0.5	2.1	8.4	4	16	0	0

Clay County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	16	0.6	2.7	5.7	5	31	0	0
First Floor Bedroom	6	0.5	1.7	5.6	1	17	0	0
Other	1	1.9	1.9	1.9	0	0	0	0
Total	23	0.5	2.4	5.7	6	26	0	0

Coles County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	35	0.6	3.7	18.5	12	34	0	0
First Floor Bedroom	1	0.8	0.8	0.8	0	0	0	0
Total	36	0.6	3.7	18.5	12	33	0	0

Table 4 (cont'd)

Cook County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	212	0.6	2.9	11.6	40	19	0	0
First Floor Bedroom	6	1	2.1	3	0	0	0	0
First Floor Living Area	43	0.5	2.3	5.8	4	9	0	0
Total	261	0.5	2.8	11.6	44	17	0	0

Crawford County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	16	0.6	2.1	9.7	1	6	0	0
First Floor Bedroom	9	0.6	0.9	1.7	0	0	0	0
First Floor Living Area	4	0.7	0.8	0.8	0	0	0	0
Other	1	0.8	0.8	0.8	0	0	0	0
Total	30	0.6	1.5	9.7	1	3	0	0

Cumberland County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	20	0.8	2.1	4.8	1	5	0	0
First Floor Bedroom	8	0.8	1.6	3.4	0	0	0	0
First Floor Living Area	7	0.8	1.0	1.6	0	0	0	0
Total	35	0.8	1.8	4.8	1	3	0	0

DeKalb County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	49	1.2	4.5	18.9	22	45	0	0
First Floor Bedroom	4	1.7	3.9	7.1	1	25	0	0
First Floor Living Area	3	1.7	2.0	2.5	0	0	0	0
Total	56	1.2	4.3	18.9	23	41	0	0

DeWitt County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	1.4	8.8	75.6	15	68	1	5
First Floor Bedroom	2	1.1	2.5	3.9	0	0	0	0
First Floor Living Area	5	0.8	1.3	1.6	0	0	0	0
Total	29	0.8	7.0	75.6	15	52	1	3

Table 4 (cont'd)

DuPage County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	134	0.5	4.8	64.5	47	35	3	2
First Floor Bedroom	13	1.1	3.1	5.4	3	23	0	0
First Floor Living Area	19	0.8	2.4	6.4	1	5	0	0
Total	166	0.5	4.4	64.5	51	31	3	2

Edgar County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	21	0.9	3.5	14.3	7	33	0	0
First Floor Bedroom	10	0.8	1.9	5	1	10	0	0
Total	31	0.8	3.0	14.3	8	26	0	0

Edwards County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.7	2.7	17.8	3	14	0	0
First Floor Bedroom	7	0.8	0.9	1.2	0	0	0	0
Total	29	0.7	2.3	17.8	3	10	0	0

Effingham County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	19	0.7	5.4	19.3	9	47	0	0
First Floor Bedroom	4	0.9	1.1	1.2	0	0	0	0
First Floor Living Area	14	0.5	1.5	3.8	0	0	0	0
Total	37	0.5	3.5	19.3	9	24	0	0

Fayette County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	28	0.9	3.0	8.4	6	21	0	0
First Floor Bedroom	11	0.9	2.8	7.1	2	18	0	0
First Floor Living Area	1	2.2	2.2	2.2	0	0	0	0
Total	40	0.9	2.9	8.4	8	20	0	0

Table 4 (cont'd)

Ford County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	17	0.1	3.7	9.1	7	41	0	0
First Floor Bedroom	8	0.8	2.7	4.5	2	25	0	0
First Floor Living Area	4	0.8	2.1	5.1	1	25	0	0
Total	29	0.1	3.2	9.1	10	34	0	0

Franklin County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	15	0.7	2.8	5.2	5	33	0	0
First Floor Bedroom	11	0.7	1.7	5.7	1	9	0	0
First Floor Living Area	8	0.8	0.9	1.1	0	0	0	0
Total	34	0.7	2.0	5.7	6	18	0	0

Fulton County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	14	1	4.9	18.1	3	21	0	0
First Floor Bedroom	14	0.9	2.2	4.7	3	21	0	0
First Floor Living Area	6	1.4	4.2	10.3	2	33	0	0
Total	34	0.9	3.7	18.1	8	24	0	0

Gallatin County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
First Floor Bedroom	31	0.9	2.4	6	4	13	0	0
Total	31	0.9	2.4	6	4	13	0	0

Greene County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	8	1	5.2	16.3	4	50	0	0
First Floor Bedroom	3	1.1	2.4	3.6	0	0	0	0
First Floor Living Area	2	0.4	0.6	0.7	0	0	0	0
Total	13	0.4	3.8	16.3	4	31	0	0

Table 4 (cont'd)

Hamilton County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	14	0.7	3.4	8.2	5	36	0	0
First Floor Bedroom	10	0.8	2.6	5.5	2	20	0	0
First Floor Living Area	6	0.8	1.1	1.6	0	0	0	0
Total	30	0.7	2.7	8.2	7	23	0	0

Hardin County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	23	1.7	4.8	13.5	9	39	0	0
First Floor Bedroom	2	0.9	2.1	3.2	0	0	0	0
First Floor Living Area	5	2.2	3.0	4.4	1	20	0	0
Total	30	0.9	4.3	13.5	10	33	0	0

Henderson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	5	1	3.0	4.6	2	40	0	0
First Floor Bedroom	2	3.1	3.5	3.8	0	0	0	0
First Floor Living Area	18	0.7	1.2	2.6	0	0	0	0
Total	25	0.7	1.7	4.6	2	8	0	0

Henry County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	37	1	7.1	25.4	27	73	2	5
First Floor Bedroom	2	6.9	8.1	9.3	2	100	0	0
First Floor Living Area	1	2.8	2.8	2.8	0	0	0	0
Total	40	1	7.1	25.4	29	72	2	5

Iroquois County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	23	0.7	3.4	16.9	4	17	0	0
First Floor Bedroom	5	0.7	1.2	1.7	0	0	0	0
First Floor Living Area	2	0.4	0.6	0.8	0	0	0	0
Total	30	0.4	2.9	16.9	4	13	0	0

Table 4 (cont'd)

Jackson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	14	0.3	1.9	4.3	1	7	0	0
First Floor Bedroom	7	0.5	0.8	1.9	0	0	0	0
First Floor Living Area	12	0.09	1.6	6.6	1	8	0	0
Other	2	3	3.0	3	0	0	0	0
Total	35	0.09	1.6	6.6	2	6	0	0

Jasper County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.4	1.7	7	2	9	0	0
First Floor Bedroom	7	0.5	0.8	1.5	0	0	0	0
Other	1	2	2.0	2	0	0	0	0
Total	30	0.4	1.5	7	2	7	0	0

Jefferson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	19	0.7	1.9	7.4	2	11	0	0
First Floor Bedroom	10	0.3	1.4	5.8	1	10	0	0
First Floor Living Area	4	0.7	1.5	2.9	0	0	0	0
Total	33	0.3	1.7	7.4	3	9	0	0

Jersey County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	6	2	3.3	8.1	1	17	0	0
First Floor Living Area	2	0.2	1.6	2.9	0	0	0	0
Total	8	0.2	2.8	8.1	1	12	0	0

Jo Daviess County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	12	1	8.8	37.5	6	50	2	17
First Floor Bedroom	2	0.6	0.6	0.6	0	0	0	0
First Floor Living Area	8	0.8	5.4	18	3	38	0	0
Total	22	0.6	6.8	37.5	9	41	2	9

Table 4 (cont'd)

Johnson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	11	1.2	3.2	5.6	3	27	0	0
First Floor Bedroom	2	2.4	2.5	2.5	0	0	0	0
First Floor Living Area	15	0.6	1.5	2.6	0	0	0	0
Total	28	0.6	2.2	5.6	3	11	0	0

Kane County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	58	0.6	6.0	34.4	30	52	1	2
First Floor Bedroom	3	1	3.8	6.8	1	33	0	0
First Floor Living Area	10	0.9	3.1	5.7	3	30	0	0
Total	71	0.6	5.5	34.4	34	48	1	1

Kankakee County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	26	0.3	3.1	16.8	8	31	0	0
First Floor Bedroom	7	0.6	1.9	5.7	1	14	0	0
First Floor Living Area	1	0.9	0.9	0.9	0	0	0	0
Other	1	0.6	0.6	0.6	0	0	0	0
Total	35	0.3	2.7	16.8	9	26	0	0

Kendall County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	25	1	6.2	19.1	16	64	0	0
First Floor Bedroom	2	1.6	2.3	3	0	0	0	0
Total	27	1	6.0	19.1	16	59	0	0

Knox County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	52	0.9	5.5	21.1	32	62	1	2
First Floor Bedroom	1	19.3	19.3	19.3	1	100	0	0
First Floor Living Area	3	2.4	7.2	13.7	2	67	0	0
Other	1	12.2	12.2	12.2	1	100	0	0
Total	57	0.9	5.9	21.1	36	63	1	2

Table 4 (cont'd)

Lake County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	79	0.4	2.3	9.6	11	14	0	0
First Floor Bedroom	10	0.5	1.9	8.9	1	10	0	0
First Floor Living Area	1	0.5	0.5	0.5	0	0	0	0
Total	90	0.4	2.3	9.6	12	13	0	0

LaSalle County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	30	1	5.0	15.5	13	43	0	0
First Floor Bedroom	1	0.3	0.3	0.3	0	0	0	0
Total	31	0.3	4.8	15.5	13	42	0	0

Lawrence County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	1	3.4	3.4	3.4	0	0	0	0
First Floor Bedroom	2	0.8	1.2	1.6	0	0	0	0
First Floor Living Area	26	0.6	1.0	5.1	1	4	0	0
Total	29	0.6	1.1	5.1	1	3	0	0

Lee County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	33	0.4	3.4	15	8	24	0	0
Other	1	1.8	1.8	1.8	0	0	0	0
Total	34	0.4	3.4	15	8	24	0	0

Livingston County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	26	1.5	7.0	39.8	18	69	1	4
First Floor Bedroom	3	3.5	6.5	12.5	1	33	0	0
First Floor Living Area	2	4.5	5.0	5.4	2	100	0	0
Total	31	1.5	6.8	39.8	21	68	1	3

Table 4 (cont'd)

Logan County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	38	1.2	6.2	19.2	24	63	0	0
First Floor Bedroom	14	0.8	2.3	3.9	0	0	0	0
First Floor Living Area	5	0.8	3.0	4.8	2	40	0	0
Total	57	0.8	5.0	19.2	26	46	0	0

Macon County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	47	0.5	3.8	15.7	14	30	0	0
First Floor Bedroom	12	0.6	1.8	3.1	0	0	0	0
First Floor Living Area	10	0.5	1.8	5.5	1	10	0	0
Other	2	0.9	1.0	1	0	0	0	0
Total	71	0.5	3.1	15.7	15	21	0	0

Macoupin County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	8	0.9	2.9	7.5	2	25	0	0
First Floor Bedroom	2	1	2.9	4.8	1	50	0	0
First Floor Living Area	19	0.8	1.8	6.8	2	11	0	0
Total	29	0.8	2.2	7.5	5	17	0	0

Madison County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	61	0.5	2.7	34.2	7	11	1	2
First Floor Bedroom	3	1.1	1.5	1.9	0	0	0	0
Other	1	0.9	0.9	0.9	0	0	0	0
Total	65	0.5	2.6	34.2	7	11	1	2

Marion County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	20	0.6	1.9	4.6	3	15	0	0
First Floor Bedroom	8	0.5	0.7	1.4	0	0	0	0
First Floor Living Area	3	0.5	0.6	0.9	0	0	0	0
Total	31	0.5	1.5	4.6	3	10	0	0

Table 4 (cont'd)

Marshall County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.9	6.8	23.4	9	41	3	14
First Floor Bedroom	4	0.8	1.5	3.4	0	0	0	0
Total	26	0.8	6.0	23.4	9	35	3	12

Mason County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	23	1.8	6.8	20.8	16	70	1	4
First Floor Bedroom	4	1.7	2.8	4.7	1	25	0	0
First Floor Living Area	2	1.6	2.0	2.3	0	0	0	0
Total	29	1.6	5.9	20.8	17	59	1	3

Massac County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	10	2.6	7.9	28.1	5	50	1	10
First Floor Bedroom	3	1.2	1.7	2.2	0	0	0	0
First Floor Living Area	13	1.3	2.6	4.4	1	8	0	0
Total	26	1.2	4.5	28.1	6	23	1	4

McDonough County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	51	0.4	5.2	17.8	25	49	0	0
First Floor Bedroom	12	0.6	2.4	8	2	17	0	0
First Floor Living Area	4	0.4	1.6	3.8	0	0	0	0
Total	67	0.4	4.5	17.8	27	40	0	0

McHenry County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	65	0.6	4.4	23.6	20	31	1	2
First Floor Bedroom	8	0.5	2.4	3.9	0	0	0	0
First Floor Living Area	3	3.2	8.1	13.2	2	67	0	0
Total	76	0.5	4.3	23.6	22	29	1	1

Table 4 (cont'd)

McLean County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	49	0.2	5.8	23.2	28	57	2	4
First Floor Bedroom	6	0.8	5.1	11.2	3	50	0	0
First Floor Living Area	3	3.1	4.4	6.4	1	33	0	0
Total	58	0.2	5.7	23.2	32	55	2	3

Menard County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	20	0.8	6.7	20	13	65	0	0
First Floor Bedroom	9	0.8	1.4	3.5	0	0	0	0
First Floor Living Area	1	0.9	0.9	0.9	0	0	0	0
Total	30	0.8	4.9	20	13	43	0	0

Mercer County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	25	2.2	9.2	23.1	17	68	3	12
First Floor Living Area	3	1.6	3.3	5.5	1	33	0	0
Total	28	1.6	8.5	23.1	18	64	3	11

Monroe County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	26	1.3	5.4	15.4	12	46	0	0
First Floor Bedroom	1	3.5	3.5	3.5	0	0	0	0
First Floor Living Area	1	2.4	2.4	2.4	0	0	0	0
Total	28	1.3	5.2	15.4	12	43	0	0

Montgomery County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.3	4.2	16.9	9	41	0	0
First Floor Bedroom	12	0.3	1.5	6	1	8	0	0
First Floor Living Area	1	1.2	1.2	1.2	0	0	0	0
Total	35	0.3	3.2	16.9	10	29	0	0

Table 4 (cont'd)

Morgan County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	41	2.2	7.1	14.6	32	78	0	0
First Floor Bedroom	17	1.2	6.4	19.2	7	41	0	0
First Floor Living Area	1	2.7	2.7	2.7	0	0	0	0
Total	59	1.2	6.8	19.2	39	66	0	0

Moultrie County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	14	0.5	3.4	11	5	36	0	0
First Floor Bedroom	13	0.5	2.1	10.8	1	8	0	0
First Floor Living Area	3	0.8	1.7	2.9	0	0	0	0
Total	30	0.5	2.6	11	6	20	0	0

Ogle County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	28	0.6	3.9	16.8	10	36	0	0
First Floor Living Area	1	1.2	1.2	1.2	0	0	0	0
Total	29	0.6	3.9	16.8	10	34	0	0

Peoria County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	46	0.7	5.6	22.4	23	50	1	2
First Floor Bedroom	4	1.1	1.9	3.3	0	0	0	0
First Floor Living Area	3	1.5	2.1	2.8	0	0	0	0
Other	2	2.3	2.6	2.8	0	0	0	0
Total	55	0.7	5.0	22.4	23	42	1	2

Perry County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	21	0.9	2.1	4.2	1	5	0	0
First Floor Bedroom	10	0.8	1.3	2.2	0	0	0	0
First Floor Living Area	4	0.9	1.1	1.5	0	0	0	0
Total	35	0.8	1.8	4.2	1	3	0	0

Table 4 (cont'd)

Piatt County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	16	0.8	4.4	17.9	5	31	0	0
First Floor Bedroom	1	1.3	1.3	1.3	0	0	0	0
First Floor Living Area	18	0.9	2.2	5.3	2	11	0	0
Total	35	0.8	3.2	17.9	7	20	0	0

Pike County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	25	0.9	5.3	15.9	12	48	0	0
First Floor Bedroom	2	0.9	1.1	1.2	0	0	0	0
First Floor Living Area	3	0.9	5.3	10	2	67	0	0
Total	30	0.9	5.0	15.9	14	47	0	0

Pope County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	17	1.2	4.7	17.6	8	47	0	0
First Floor Bedroom	3	1.5	2.2	3.4	0	0	0	0
First Floor Living Area	10	1.6	2.5	4.8	1	10	0	0
Total	30	1.2	3.8	17.6	9	30	0	0

Pulaski County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	6	1.9	2.9	4	0	0	0	0
First Floor Bedroom	16	1.4	2.3	3.2	0	0	0	0
First Floor Living Area	5	1.5	1.7	2.1	0	0	0	0
Total	27	1.4	2.3	4	0	0	0	0

Putnam County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	18	0.8	5.7	43.8	6	33	1	6
First Floor Bedroom	5	0.9	1.9	4.3	1	20	0	0
First Floor Living Area	2	0.4	1.9	3.3	0	0	0	0
Other	1	0.7	0.7	0.7	0	0	0	0
Total	26	0.4	4.5	43.8	7	27	1	4

Table 4 (cont'd)

Randolph County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	16	0.5	2.1	4.7	2	12	0	0
First Floor Bedroom	8	0.7	2.0	4.8	2	25	0	0
First Floor Living Area	12	0.5	1.2	4.4	1	8	0	0
Total	36	0.5	1.8	4.8	5	14	0	0

Richland County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.8	2.3	5.3	3	14	0	0
First Floor Bedroom	7	0.9	1.1	1.8	0	0	0	0
Total	29	0.8	2.0	5.3	3	10	0	0

Rock Island County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	63	0.5	6.2	46.2	34	54	2	3
First Floor Bedroom	1	2.1	2.1	2.1	0	0	0	0
First Floor Living Area	1	3.5	3.5	3.5	0	0	0	0
Other	1	0.6	0.6	0.6	0	0	0	0
Total	66	0.5	6.0	46.2	34	52	2	3

Saline County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	8	0.4	1.7	3.9	0	0	0	0
First Floor Bedroom	14	0.4	1.9	4.9	1	7	0	0
First Floor Living Area	8	0.4	2.1	4.3	1	12	0	0
Total	30	0.4	1.9	4.9	2	7	0	0

Sangamon County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	72	1.2	4.5	22.4	29	40	1	1
First Floor Bedroom	23	0.9	1.9	3.3	0	0	0	0
First Floor Living Area	8	0.9	4.6	23.2	1	12	1	12
Total	103	0.9	3.9	23.2	30	29	2	2

Table 4 (cont'd)

Schuyler County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	23	0.8	5.4	19	11	48	0	0
First Floor Bedroom	6	0.8	4.8	8.9	3	50	0	0
Total	29	0.8	5.3	19	14	48	0	0

Scott County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	18	1.4	6.2	10.9	13	72	0	0
First Floor Bedroom	11	2	3.4	6.5	3	27	0	0
First Floor Living Area	1	3.4	3.4	3.4	0	0	0	0
Total	30	1.4	5.1	10.9	16	53	0	0

Shelby County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	0.7	3.2	10.2	6	27	0	0
First Floor Bedroom	5	0.7	1.9	3.2	0	0	0	0
First Floor Living Area	1	2.1	2.1	2.1	0	0	0	0
Total	28	0.7	3.0	10.2	6	21	0	0

St. Clair County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	40	0.4	3.2	23	7	18	1	2
First Floor Bedroom	5	0.8	3.5	6.4	2	40	0	0
First Floor Living Area	5	0.4	1.0	1.4	0	0	0	0
Total	50	0.4	3.0	23	9	18	1	2

Stark County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	27	0.7	6.8	21.3	17	63	1	4
First Floor Living Area	1	1.9	1.9	1.9	0	0	0	0
Total	28	0.7	6.7	21.3	17	61	1	4

Table 4 (cont'd)

Stephenson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	53	0.7	4.5	14.9	27	51	0	0
First Floor Living Area	1	1.4	1.4	1.4	0	0	0	0
Other	1	4.8	4.8	4.8	1	100	0	0
Total	55	0.7	4.5	14.9	28	51	0	0

Tazwell County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	48	0.9	5.9	14.3	33	69	0	0
First Floor Bedroom	6	1.5	4.6	11.3	3	50	0	0
First Floor Living Area	5	0.9	3.1	8.8	1	20	0	0
Total	59	0.9	5.5	14.3	37	63	0	0

Union County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	22	1.1	3.4	5.9	8	36	0	0
First Floor Bedroom	7	1.3	2.1	2.9	0	0	0	0
First Floor Living Area	1	2	2.0	2	0	0	0	0
Total	30	1.1	3.1	5.9	8	27	0	0

Vermillion County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	28	0.9	4.4	12.8	11	39	0	0
First Floor Bedroom	4	0.9	1.8	3	0	0	0	0
First Floor Living Area	4	0.7	2.4	5.6	1	25	0	0
Total	36	0.7	3.9	12.8	12	33	0	0

Wabash County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	17	0.8	1.6	2.9	0	0	0	0
First Floor Bedroom	12	0.7	1.1	2.6	0	0	0	0
First Floor Living Area	2	0.7	2.0	3.2	0	0	0	0
Other	1	5.5	5.5	5.5	1	100	0	0
Total	32	0.7	1.6	5.5	1	3	0	0

Table 4 (cont'd)

Warren County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	42	2.2	12.4	59.5	37	88	8	19
First Floor Bedroom	4	4.1	6.9	9.4	4	100	0	0
First Floor Living Area	3	2.6	5.0	9.3	1	33	0	0
Total	49	2.2	11.5	59.5	42	86	8	16

Washington County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	34	0.2	2.4	7.2	5	15	0	0
First Floor Bedroom	2	0.8	0.8	0.8	0	0	0	0
Total	36	0.2	2.4	7.2	5	14	0	0

Wayne County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	11	0.3	1.3	3.1	0	0	0	0
First Floor Bedroom	5	0.6	0.7	0.9	0	0	0	0
First Floor Living Area	3	0.3	0.6	1.1	0	0	0	0
Total	19	0.3	1.0	3.1	0	0	0	0

White County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	20	0.4	2.5	6.4	4	20	0	0
First Floor Bedroom	12	0.4	1.3	2.3	0	0	0	0
First Floor Living Area	1	1	1.0	1	0	0	0	0
Total	33	0.4	2.0	6.4	4	12	0	0

Whiteside County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	33	0.7	2.9	8.1	7	21	0	0
First Floor Bedroom	3	0.9	2.0	3.5	0	0	0	0
Total	36	0.7	2.8	8.1	7	19	0	0

Table 4 (cont'd)

Will County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	47	0.5	3.9	16.8	14	30	0	0
First Floor Bedroom	8	0.6	2.3	4.1	1	12	0	0
First Floor Living Area	9	0.6	1.3	4.7	1	11	0	0
Total	64	0.5	3.3	16.8	16	25	0	0

Williamson County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	15	0.8	3.6	11.7	3	20	0	0
First Floor Bedroom	13	0.7	1.2	2.4	0	0	0	0
First Floor Living Area	7	0.8	0.9	1.3	0	0	0	0
Total	35	0.7	2.2	11.7	3	9	0	0

Winnebago County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	52	1.2	4.1	19	19	37	0	0
First Floor Bedroom	4	1.7	2.5	4.3	1	25	0	0
First Floor Living Area	2	1.9	2.3	2.6	0	0	0	0
Total	58	1.2	3.9	19	20	34	0	0

Woodford County

	Number	Min Result	Avg Result	Max Result	#>4 pCi/L	%>4 pCi/L	#>20 pCi/L	%>20 pCi/L
Basement	26	1	10.2	33.7	20	77	3	12
First Floor Bedroom	2	2.8	5.9	9	1	50	0	0
Total	28	1	9.9	33.7	21	75	3	11

Table 5

SUMMARY OF ILLINOIS SCHOOL RADON SCREENING RESULTS

66 SCHOOLS
23 COUNTIES

	Number	Min Result	Max Result	%>4 pCi/L	%>20 pCi/L
Basement	51	0.1*	25.8	12	2
First Floor	577	0.1*	10.0	3	0
Second Floor	52	0.5*	8.0	6	0
Third Floor	23	0.7*	5.0	4	0

*Less than Minimum Detectable Concentration

Table 6

CORRELATION BETWEEN RADON CONCENTRATIONS AND BUILDING CHARACTERISTICS

Age of House	Number	Average (pCi/L)
Less than 15 years old	919	3.9
Greater than 50 years old	1388	4.1

Substructure Type	Number	Average (pCi/L)
100% Basement	1760	4.1
100% Slab	164	3.4
100% Crawlspace	535	2.0
Basement and Slab	223	5.1
Basement and Crawl Space	880	4.6

Subjective Energy Efficiency	Number	Average (pCi/L)
Not at all	184	3.2
Somewhat	37	2.5
Adequate	333	3.6
Good	1302	3.9
Excellent	2272	4.0

Basement Characteristics	Number	Average (pCi/L)
Exposed Earth	239	5.3
Sump(s)	885	4.5
Crack(s)	784	4.6
Drain(s)	1660	4.6
None of the above	50	3.5
All of the above	39	5.6

Crawlspace Characteristics	Number	Average (pCi/L)
Crawlspace Entry & Exposed Earth	480	4.7
Crawlspace Vented	504	3.1

Primary Heating Source	Number	Average (pCi/L)
Solar	5	7.5
Oil	174	4.8
Electric	421	3.8
Natural Gas	2689	4.0
Propane	448	3.8
Wood	174	3.0
Coal	6	1.5

Other Factors	Number	Average (pCi/L)
Central Air Conditioning	1367	4.2

Table 7

SCHOOL RADON SCREENING RESULTS
pCi/L

County	Basement		1st Floor		2nd Floor		3rd Floor	
	n	Range	n	Range	n	Range	n	Range
Calhoun	3	2.1-3.8	13	1.1-3.3	2	0.5*-1.9	0	
Champaign	2	3.2-4.5	3	1.7-4.2	3	1.4-3.7	2	0.8*-1.2
Clark	0		204	0.1*-4.3	0		0	
DeWitt	2	3.4-3.9	6	1.4-3.2	2	2.0-2.8	0	
Effingham	0		4	0.8*-1.2	4	0.8*-1.2	0	
Ford	1	4.6	7	1.5*-2.9	2	0.8*-1.4	2	0.7*-2.4
Gallatin	0		4	1.3-2.1	2	1.4-1.7	2	1.4-1.5
Henry	0		8	1.2*-10.0	2	0.8*-2.2	2	1.1-1.5
LaSalle	1	2.3	10	0.8*-2.2	1	0.8*	0	
Livingston	0		7	0.7*-1.5	2	1.9-3.8	2	0.7*-0.7*
McLean	0		5	4.3-9.2	5	3.3-8.0	2	3.2-5.0
Monroe	0		6	0.9-3.0	6	1.0-2.7	0	
Montgomery	0		8	1.6-3.2	2	1.7-1.8	2	1.0-1.5
Moultrie	0		2	2.3-4.5	2	1.0-1.2	2	1.7-1.8
Pike	2	2.3-6.0	15	0.2*-5.8	1	1.3	0	

Table 7 (cont'd)

SCHOOL RADON SCREENING RESULTS
pCi/L

County	Basement		1st Floor		2nd Floor		3rd Floor	
	n	Range	n	Range	n	Range	n	Range
Saline	3	1.5-4.4	3	0.7-1.6	3	0.7*-1.4	0	
Sangamon	1	25.8	1	1.9	1	3.1	0	
Schuyler	0		8	1.1-6.3	2	1.1-2.2	2	1.5-1.7
St. Clair	0		6	1.6-3.1	0		0	
Wayne	36	0.1*-1.4	241	0.1*-3.6	0		0	
White	0		4	0.7-1.6	4	0.7-2.2	0	
Will	0		8	0.9*-2.3	2	0.5*-0.9*	2	1.4-1.4
Woodford	0		4	0.8*-5.6	4	1.0-3.4	3	1.2-2.7

* Less Than Minimum Detectable Concentration
n = Number of measurements

ANDREWS OFFICE PRODUCTS CAPITOL HEIGHTS, MD (K)

RESEARCH TRIANGLE INSTITUTE

July 16, 1992 RTI/5158/49-1F

FINAL REPORT

NATIONAL RESIDENTIAL RADON SURVEY
STATISTICAL ANALYSIS

VOLUME 1
NATIONAL AND REGIONAL ESTIMATES

Prepared for

Office of Radiation Programs
U.S. Environmental Protection Agency
Washington, D.C. 20460

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Under Subcontract to

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McLean, Virginia 22101

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Work Assignment 2-49

Table 2-1

Definition of the Twenty-Two Design Strata

EPA Region	Expected Radon Level	Stratum ID Number	State/Substate Area
Region 1:	High:	013	ME, NH, VT.
	Medium	012	MA, CT, RI.
Region 2:	High:	023	Northern NJ.
	Medium	022	NY.
	Low:	021	Southern NJ.
Region 3:	High:	033	PA, Western MD, WV, Western VA.
	Low:	031	DE, Central and Eastern VA, Eastern MD, DC
Region 4:	High:	043	Western NC, Western SC, Northern GA, Northern AL, Eastern TN.
	Medium:	042	KY, Western and Central TN.
	Low:	041	Central and Eastern NC, Eastern SC, Southern GA, Southern AL, MS, FL.
Region 5:	High:	053	MN, WI, IL, IN, OH.
	Low:	051	MI.
Region 6:	High:	063	NM
	Medium:	062	OK, Western and Central TX, Northern AR.
	Low:	061	LA, Southern AR, Southeastern TX.
Region 7:	High:	073	NE, IA.
	Medium:	072	KS, MO.
Region 8:	High:	083	MT, WY, UT, CO, ND, SD.
Region 9:	High:	093	NV.
	Low:	091	CA, AZ, HI.
Region 10:	High:	103	AK, ID.
	Low:	101	WA, OR.

Table 3-1
Estimated Percent of Year-Round Occupied Housing Units with Annual-Average Radon Concentrations above Key Thresholds: United States, 1989-1990, page 1 of 3

EPA Region	Lowest Level of Living Space: Percent > 4 pCi/L	Lowest Level of Living Space: Percent > 10 pCi/L	Lowest Level of Nonliving Space(1): Percent > 4 pCi/L	Lowest Level of Nonliving Space(1): Percent > 10 pCi/L	Average Over All Living Levels: Percent > 4 pCi/L	Average Over All Living Levels: Percent > 10 pCi/L	Weighted Average Over All Levels(2): Percent > 4 pCi/L	Weighted Average Over All Levels(2): Percent > 10 pCi/L
United States								
Percent	8.01	1.28	25.90	5.68	6.01	0.65	5.16	0.43
(Std. Err.)	(0.81)	(0.27)	(2.74)	(1.06)	(0.68)	(0.18)	(0.66)	(0.15)
No. of Obs.	584	91	428	92	439	51	1,027	89
Region I								
Percent	10.01	0.66	23.97	7.32	4.12	0.50	3.88	0.29
(Std. Err.)	(3.68)	(0.49)	(7.53)	(2.84)	(1.14)	(0.40)	(0.87)	(0.20)
No. of Obs.	38	3	41	13	19	2	56	5
Region II								
Percent	6.20	1.68	20.86	6.18	3.91	0.72	2.32	0.37
(Std. Err.)	(2.22)	(0.58)	(8.30)	(3.48)	(1.34)	(0.35)	(1.26)	(0.26)
No. of Obs.	35	10	33	10	23	4	39	6
Region III								
Percent	10.04	2.43	17.56	4.81	7.36	1.40	7.09	1.28
(Std. Err.)	(2.80)	(1.56)	(4.11)	(3.06)	(2.17)	(1.04)	(2.76)	(1.12)
No. of Obs.	89	20	51	16	63	11	152	27

Notes: (1) Residences that used every level as living space are excluded from this column.

(2) This column includes one observation for every occupant of the 5,694 final responding households.

(3) The entry N.A. indicates a radon concentration category with no observations, giving an estimate of 0.00 percent. Although some residences in the Region may have radon concentrations in the range, none were observed. This suggests that the percent of homes in the Region with radon concentrations in the range is small. The few number of observations in some Regions also reduces the precision of the estimates for the other radon concentration categories. Estimates with a relative standard error greater than 50 percent of the point estimate should be interpreted with caution.

Table 3-1
Estimated Percent of Year-Round Occupied Housing Units with Annual-Average Radon Concentrations above Key Thresholds: United States, 1989-1990, page 2 of 3

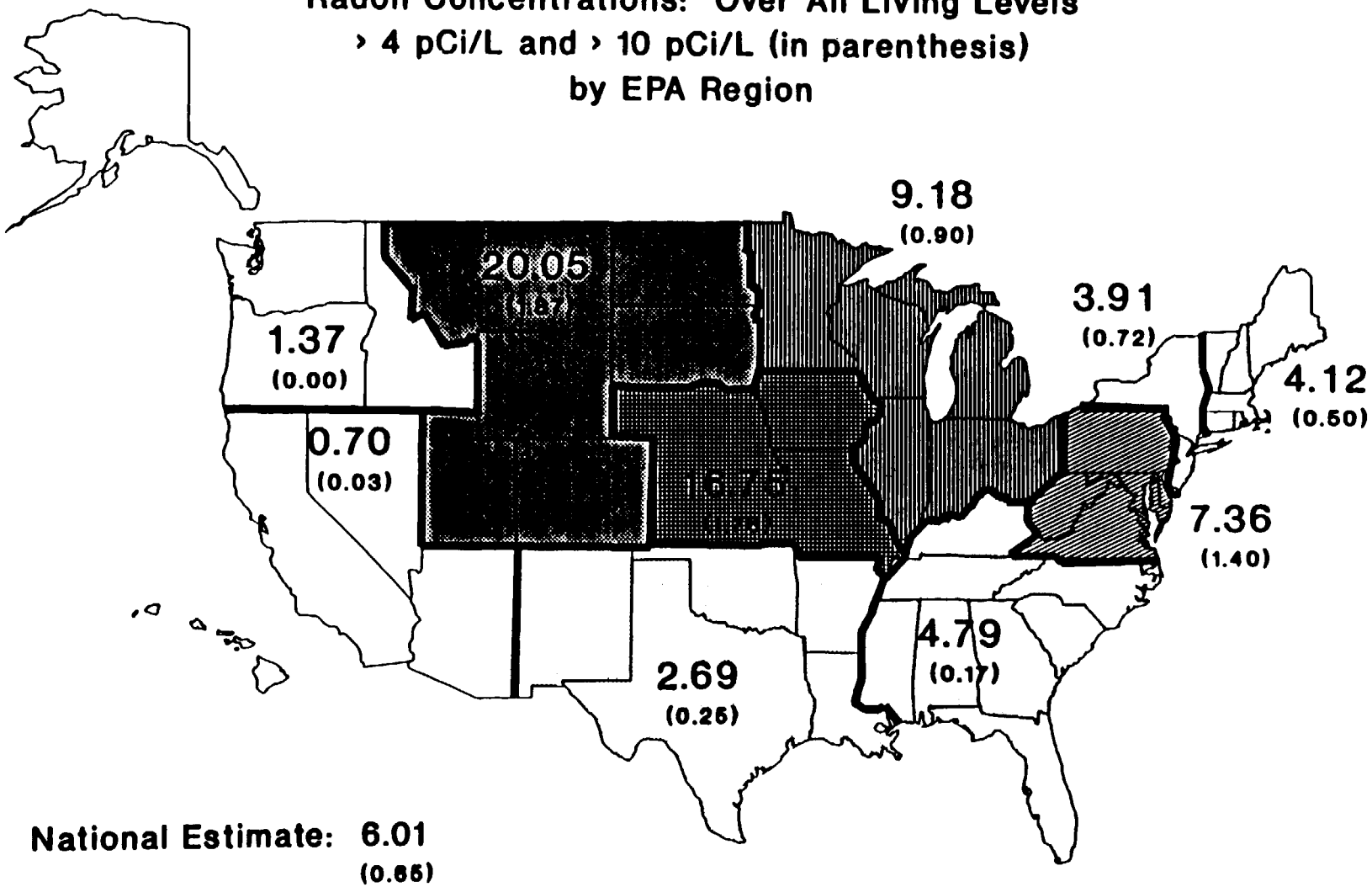
EPA Region	Lowest Level of Living Space: Percent > 4 pCi/L	Lowest Level of Living Space: Percent > 10 pCi/L	Lowest Level of Nonliving Space(1): Percent > 4 pCi/L	Lowest Level of Nonliving Space(1): Percent > 10 pCi/L	Average Over All Living Levels: Percent > 4 pCi/L	Average Over All Living Levels: Percent > 10 pCi/L	Weighted Average Over All Levels(2): Percent > 4 pCi/L	Weighted Average Over All Levels(2): Percent > 10 pCi/L
Region IV								
Percent	5.59	0.96	27.85	5.38	4.79	0.17	4.47	0.09
(Std. Err.)	(1.64)	(0.56)	(7.65)	(3.20)	(1.78)	(0.09)	(1.66)	(0.04)
No. of Obs.	56	6	31	6	45	2	113	3
Region V								
Percent	12.21	1.42	26.21	3.55	9.18	0.90	7.70	0.49
(Std. Err.)	(1.60)	(0.43)	(2.19)	(0.83)	(1.56)	(0.40)	(1.36)	(0.22)
No. of Obs.	197	25	161	20	149	16	343	25
Region VI								
Percent	2.93	0.25	0.00	0.00	2.69	0.25	2.29	0.19
(Std. Err.)	(1.02)	(0.24)	N.A.(3)	N.A.	(1.02)	(0.24)	(0.84)	(0.19)
No. of Obs.	18	1	0	0	17	1	37	2
Region VII								
Percent	21.08	3.72	35.29	6.76	16.75	1.76	14.65	0.85
(Std. Err.)	(8.45)	(2.32)	(11.98)	(3.11)	(6.94)	(1.73)	(5.80)	(0.63)
No. of Obs.	69	15	33	6	55	8	134	9

Notes: (1) Residences that used every level as living space are excluded from this column.

(2) This column includes one observation for every occupant of the 5,694 final responding households.

(3) The entry N.A. indicates a radon concentration category with no observations, giving an estimate of 0.00 percent. Although some residences in the Region may have radon concentrations in the range, none were observed. This suggests that the percent of homes in the Region with radon concentrations in the range is small. The few number of observations in some Regions also reduces the precision of the estimates for the other radon concentration categories. Estimates with a relative standard error greater than 50 percent of the point estimate should be interpreted with caution.

Figure 3-1 Percent of Housing Units with Annual-Average Radon Concentrations: Over All Living Levels › 4 pCi/L and › 10 pCi/L (in parenthesis) by EPA Region



National Estimate: 6.01
(0.65)

Note: Standard errors for the regional estimates are given in Table 3-1. Estimates with relative standard errors greater than 50 percent of the point estimate should be interpreted with caution.

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Paper

INDOOR ^{222}Rn CONCENTRATIONS IN A PROBABILITY SAMPLE OF 43,000 HOUSES ACROSS 30 STATES

S. B. White,* Jane W. Bergsten,* Barbara V. Alexander,* Nathaniel F. Rodman* and Jeffrey L. Phillips†

Abstract—**The U.S. Environmental Protection Agency has assisted 30 of the 48 conterminous states in completing statistically designed surveys of indoor ^{222}Rn over the past 4 y. In all states, the lowest livable level of 43,054 randomly selected houses was tested using charcoal canisters exposed for 48 h. The sampled population included owner-occupied ground-level houses having listed telephone numbers. Summary statistics along with the percentage of houses exceeding various concentration levels are given by state and over all states for houses with basements, for houses without basements, and for all houses. As expected, ^{222}Rn concentration varies widely from one state to another and, in every state, basement houses exhibit higher concentrations than nonbasement houses. The lognormal distribution is shown to be a good approximation to the distribution of screening measurements over the 30-state area. There is, however, some evidence that the lognormal distribution underestimates, by a narrow margin, the upper tail of the observed distribution of basement measurements.**

Health Phys. 62(1):41–50; 1992

Key words: ^{222}Rn; surveys; sampling; radioactivity, natural

INTRODUCTION

In an attempt to characterize the magnitude of maximum indoor ^{222}Rn concentrations across the U.S., the U.S. Environmental Protection Agency (EPA) provides assistance to states in the selection and testing of a probability-based sample of houses. The use of probabilities in making house selections allows results to be validly extrapolated beyond the sample itself to a well-defined population and enables precision of estimates to be quantified. Assistance is provided in survey design, interviewer training, sample selection, data processing, and data analysis. In addition, the Agency provides charcoal canisters used in all sample houses and also provides all laboratory analyses.

Each state survey had two objectives. The first was to characterize the distribution of maximum indoor ^{222}Rn concentrations across the state through the use of short-term tests. The second was to identify areas within the state with elevated ^{222}Rn levels. Jointly, these objectives imply a need for estimating mean concentrations and percentages of households with readings above specified concentrations for the state as a whole and for geographic regions within the state. The objectives also imply a need for testing a large number of households that are widely dispersed across the state.

Since the beginning of the state assistance program in 1986, 30 of the 48 conterminous states have completed statistically designed surveys of indoor ^{222}Rn.‡ In this group of 30 states, shown in Fig. 1, 2-d screening measurements were made in approximately 43,000 randomly selected houses. Although other surveys have used probability sampling, e.g., New York (Perritt et al. 1990) and New Jersey (Rahman et al. 1988), and other data sets include more test houses (Alter and Oswald 1987; Cohen and Gromicko 1988), this is, by far, the largest data base formed from studies that 1) use probabilities in making house selections; 2) have common objectives; 3) utilize the same measurement method; 4) employ the same protocol; and 5) sample the same target population. This paper updates and expands the work reported by Dziuban et al. (1990) by including more state surveys, combining results across states, and assessing distributional characteristics.

The surveyed states were not selected on the basis of probability but rather on the basis of their expressed interest in participating in the assistance program. Thus, the aggregated results shown herein are, from a statistical viewpoint, restricted to the 30-state area. Because of the geographic coverage of the United States (Fig. 1), it seems reasonable, however, to view the results of the 30-state area as representative of the nation as a whole.

Radon concentrations at the national level have been reported by Nero et al. (1986), Cohen (1986), and Alter and Oswald (1987). Nero et al. (1986) aggregated a number of data sets totalling less than 1000 households. These data sets differed with respect to the measurement technique, the length of the measurement period, and the time of the year measurements were

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0017-9078/92/$3.00/0

‡ Alaska and Hawaii also conducted statistically designed surveys but are excluded here because they are not members of the 48 conterminous states.

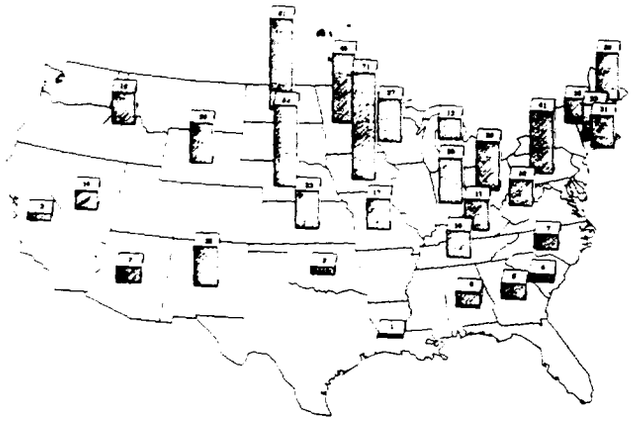

Fig. 1. Estimated percentage of houses with screening measurements >148 Bq m^{-3}.

made. Cohen (1986) tested about 500 houses of physics professors from some 100 universities in 42 states. Alpha-track detectors were exposed for 1 y in all houses. Alter and Oswald (1987) used a data base comprised of some 60,000 measurements generated by users of etched-track detectors. The period of exposure ranged from 1 wk to 1 y. The data base contains multiple measurements in the same house (at the same time or at different times) and a disproportionate number of measurements in known "hot spots" (for example, in Pennsylvania, New Jersey, and Maine). Later, some results of these studies will be contrasted to findings from the 30-state surveys.

Table 1. Disposition of sample selections.

State	Of all selections: Percent found to be survey-eligible	Of all known survey eligibles: Percent accepting a detector	Of all known survey eligibles: Percent returning a detector with a usable reading
AL	55	77	66
AZ	35	93	72
CA	44	91	69
GA	45	91	68
IA	58	92	82
ID	60	92	73
IN	44	88	76
KS	56	86	70
KY	57	71	63
LA	34	86	62
MA	48	86	70
ME	52	93	79
MI	56	65	54
MN	63	95	86
MO	58	87	72
NC	58	89	74
ND	51	91	79
NE	58	92	84
NM	52	95	81
NV	47	90	70
OH	48	89	66
OK	56	94	77
PA	59	82	68
RI	51	80	62
SC	55	91	64
TN	59	76	73
VT	48	91	74
WI	56	85	76
WV	57	89	78
WY	50	96	81
Total	52	88	72

METHODS

Each state employed 2-d screening measurements made in the lowest livable area during closed-house conditions in the winter (Ronca-Battista et al. 1988). The lowest livable area is one that could be considered livable with, at most, some cosmetic changes; if the area was in a basement, it was required to have a sealed floor. This type of screening measurement provides an estimate of the maximum concentration to which occupants can be exposed. The method was used in the state surveys because it is the one the EPA and the Centers for Disease Control currently recommend to homeowners for determining whether additional tests are needed (U.S. EPA and CDC 1986).

Different types of follow-up tests are recommended for screening measurements between 148 Bq m^{-3} (4 pCi L^{-1}) and 740 Bq m^{-3} (20 pCi L^{-1}), and above 740 Bq m^{-3}. These state surveys provide direct estimates of how many houses in the target population fall into these categories. Data from these state surveys should not, however, be used directly in assessing exposures or health risks because the measurements do not yield annual average concentrations in living areas. To use these data for exposure or risk assessment, adjustments are necessary to translate these short-term measurements into long-term estimates of average concentrations. Studies of the relationship between short-term and long-term measurements of ^{222}Rn are underway in a subset of states to quantify these adjustments.

The target population, or population to which the sample results apply, consists of all owner-occupied private or main-residence homes with (1) a listed telephone number; (2) a permanent foundation; and (3) at least one floor at or below grade level (this includes houses built over a crawl space).

For each state, a probability sample of listed residential telephone numbers was selected from a sampling frame constructed from telephone directories for all communities in the state. A probability sample is one in which every element in the sample (in this case, listed residential telephone numbers) has a positive and known chance of being included (Hansen et al. 1953; Kish 1965; Cochran 1971).

Each state was divided into strata based on population density, potential for high ^{222}Rn, and geographic areas for which separate statistical estimates were desired. Each stratum was then sampled at a different rate

Table 2. Number of houses tested, size of population sampled, and percent with basement.

State	Number of houses tested			Number of houses in target population (in 1000s)			Percentage of houses with a basement
	Basement	Nonbasement	Total[a]	Basement	Nonbasement	Total[a]	
AL	172	947	1,180	103	431	566	19.3
AZ	45	1,355	1,507	13	435	482	2.9
CA	96	1,727	1,885	99	2,055	2,233	4.6
GA	299	1,207	1,534	205	605	826	25.3
IA	1,208	166	1,381	527	64	594	89.1
ID	447	627	1,142	91	89	187	50.4
IN	997	854	1,914	492	470	993	51.1
KS	1,223	722	2,009	311	181	509	63.2
KY	373	475	879	257	308	586	45.4
LA	4	1,229	1,314	1	404	432	0.2
MA	1,420	128	1,659	865	78	1,010	91.7
ME	656	158	839	186	43	237	81.3
MI	1,591	386	1,989	1,241	269	1,520	82.2
MN	766	153	919	821	146	966	84.9
MO	1,155	687	1,859	645	344	999	65.2
NC	332	911	1,290	219	854	1,115	20.4
ND	1,371	223	1,596	167	27	194	85.8
NE	1,436	559	2,027	241	65	311	78.7
NM	127	1,715	1,887	10	177	191	5.4
NV	306	1,196	1,562	9	80	93	9.8
OH	1,246	457	1,734	1,377	436	1,844	76.0
OK	86	1,461	1,637	25	480	538	5.0
PA	2,121	248	2,389	2,004	241	2,262	89.3
RI	348	28	376	153	12	166	92.6
SC	98	917	1,089	46	425	505	9.7
TN	452	1,248	1,773	172	533	742	24.4
VT	425	271	710	71	44	118	61.8
WI	1,014	106	1,191	831	55	934	93.8
WV	464	491	1,006	145	162	324	47.3
WY	490	230	777	48	21	74	69.9

[a] Includes houses for which basement/nonbasement classification was unknown.

'for example, a sparsely settled stratum with a high potential for ^{222}Rn was oversampled). Use of this procedure ensured a wide dispersement of the sample across the state and enhanced the chances of finding areas with elevated radon. In the 30 states surveyed, 1,946 counties out of a total of 1,976 counties had at least one house tested, and 1,691 counties had at least four houses tested.

Multiple systematic random samples of telephone numbers were selected from each stratum. After selection, each state sample was partitioned into sample "waves," each consisting of a random subsample of 50 residential telephone listings. The waves were numbered sequentially and implemented in that order. This procedure of implementing the sample in waves provided each state with the option of stopping its survey at the completion of any sample wave while maintaining the capability to generate estimates and standard errors based on a probability sample.

Starting with the first wave and proceeding sequentially from wave to wave, telephone calls were made to each of the 50 sample residential phone numbers in a wave. The interviewer first screened for survey eligibility, which required that the dwelling qualify as a member of the target population described earlier. Once survey eligibility was established, the owner-occupant was given descriptive information about ^{222}Rn and the planned survey and was then asked to participate. Those agreeing to participate were mailed a charcoal canister with instructions for placing it on the lowest livable level of their home. Participants were to mail it to the EPA analysis laboratory after 48 h of exposure under closed-house conditions.

Open-face charcoal canisters (Gray and Windham 1987) were used in the first 22 states surveyed. Then the EPA switched to barrier charcoal canisters (Gray and Windham 1990) for California, Idaho, Louisiana, Nebraska, Nevada, North Carolina, Oklahoma, and South Carolina. Approximately 95% of all canisters were exposed during the months of November through April. The limit of detection is 19 Bq m^{-3} (0.5 pCi L^{-1}) for open-face canisters and 37 Bq m^{-3} (1.0 pCi L^{-1}) for barrier canisters. The performance of the charcoal canisters was monitored periodically through the use of unexposed canisters, canisters exposed to known levels of ^{222}Rn, and collocated canisters.

The number of telephone numbers initially selected for a given stratum was several times the number needed. This was necessary to compensate for telephone numbers that did not yield a survey-eligible housing

Table 3. Statewide estimates of arithmetic mean, geometric mean, and geometric standard deviation.

	Arithmetic mean (Bq m^{-3})			Geometric mean (Bq m^{-3})			Geometric standard deviation		
State	Basement	Nonbasement	All houses	Basement	Nonbasement	All houses	Basement	Nonbasement	All houses
AL	165	44	66	67	25	30	3.0	2.7	2.9
AZ	67	56	59	43	37	36	3.1	2.8	2.8
CA	67	33	33	33	19	19	3.4	3.9	3.9
GA	107	52	67	82	37	45	2.1	2.4	2.5
IA	348	204	329	241	134	226	2.5	2.6	2.6
ID	189	74	130	104	41	63	2.9	3.7	3.7
IN	181	93	137	113	54	78	2.7	2.8	3.0
KS	144	59	115	101	38	70	2.4	2.7	2.8
KY	148	58	98	81	33	49	3.0	2.9	3.2
LA	—[a]	19	19	—[a]	11	11	—[a]	3.4	3.4
MA	133	85	126	77	36	71	2.7	3.8	2.9
ME	170	63	152	100	38	83	2.8	2.7	3.0
MI	89	41	78	58	23	49	2.3	2.8	2.6
MN	185	126	178	143	76	130	2.1	2.9	2.3
MO	115	67	96	76	40	61	2.4	2.7	2.6
NC	144	30	52	70	15	22	3.0	3.7	4.0
ND	281	126	259	202	85	178	2.2	2.5	2.4
NE	229	111	204	178	70	144	2.1	3.0	2.4
NM	215	111	115	144	78	78	2.4	2.4	2.4
NV	233	52	74	126	30	37	3.7	3.4	3.7
OH	181	93	159	93	49	80	3.0	3.3	3.2
OK	100	37	41	56	19	19	3.3	3.8	3.8
PA	303	126	285	131	63	121	3.5	3.4	3.5
RI	122	78	118	75	34	70	2.6	3.1	2.7
SC	137	30	41	67	15	19	3.6	3.8	4.0
TN	198	68	99	100	41	50	3.0	2.8	3.1
VT	122	48	93	67	27	47	2.9	3.0	3.2
WI	131	46	127	87	28	80	2.5	3.1	2.8
WV	141	59	96	88	33	52	2.7	2.7	3.0
WY	156	78	131	103	50	82	2.5	2.6	2.7

[a] Not estimated because of small sample size—only four houses.

unit and for some eligible housing units that did not participate in the survey. Overall, slightly over half the sample telephone numbers yielded an eligible residence, but the percentage varied from a low of 34% for Louisiana to a high of 63% for Minnesota (Table 1). On average, about 22% of the sample telephone numbers yielded residences that were not survey eligible and about 9% yielded nonresidences. For about 16% of the telephone numbers, eligibility was never determined, principally because of persistent ring-no-answer. For the known survey-eligible residences, about 88% of the respondents completed the interview and agreed to place a canister in the home. Not all of these respondents placed and returned the canister in a timely fashion, however. Usable readings were obtained from only about 72% of the known survey-eligible homes.

Because telephone numbers in different strata were selected at different sampling rates, it was necessary to assign sampling weights that counterbalanced the unequal selection probabilities. The weights assigned were the inverse of the sample selection probabilities. An additional weight adjustment was made to compensate for nonresponse so population aggregates could be easily estimated from the sample data. All data analyses were carried out using properly weighted data that reflected the full complexity of the sample design. This permitted the generation of unbiased statistical estimates (Shah 1984).

RESULTS AND DISCUSSION

In all state surveys, the 2-d screening measurement was taken in the basement if it was considered livable; otherwise the measurement was made on the first floor. Separate analysis results are presented in this paper for houses with basements, for houses without basements and for all houses combined.

Based on program needs and available resources each state determined how many houses would be tested. The actual numbers of houses that provided valid test data are shown in Table 2 and ranged from 376 in Rhode Island to 2389 in Pennsylvania. In 24 of the 30 states, more than 1000 houses were tested. Table 2 also shows, for each state, the number of basement and nonbasement houses tested, the estimated number of houses in the target population, and the estimated percentage of houses with livable basements. The percentage of basement houses varied from 0.2% in Louisiana to 93.8% in Wisconsin and definitely reflected geographic location.

Each state survey was designed so that some houses would have a better chance of being selected into

Table 4. Statewide distributions of basement screening measurements.

State	Estimated percentage of houses with screening measurements >X Bq m^{-3} (pCi L^{-1}) for X equal to												Maximum value (Bq m^{-3})
	37 (1)	74 (2)	111 (3)	148 (4)	185 (5)	222 (6)	259 (7)	296 (8)	370 (10)	444 (12)	555 (15)	740 (20)	
AL	70.2	45.1	30.2	22.3	12.9	9.0	7.7	5.9	4.1	2.2	1.9	1.3	6,660
AZ	55.0	38.9	20.5	11.8	2.3	1.1	0.0	0.0	0.0	0.0	0.0	0.0	229
CA	58.2	19.6	10.4	7.7	6.9	6.2	2.5	2.2	1.9	1.9	1.9	0.4	1,077
GA	85.8	53.3	31.8	19.4	13.6	10.1	5.4	4.7	2.9	2.3	0.8	0.0	692
IA	97.0	90.0	82.1	73.5	65.7	59.1	51.5	43.7	33.7	24.8	15.7	8.1	4,814
ID	83.6	63.8	44.1	29.1	23.0	20.9	17.7	16.1	13.7	10.6	5.3	3.4	2,194
IN	85.8	66.9	51.4	39.2	29.4	22.4	18.4	15.3	10.8	8.4	6.0	2.3	2,657
KS	87.1	63.8	46.8	32.4	22.5	17.2	12.7	9.2	5.3	3.7	2.7	1.1	1,776
KY	74.8	49.9	33.6	28.2	22.9	19.1	16.2	13.4	8.4	6.4	4.0	3.3	2,424
LA[a]													
MA	76.5	49.1	33.6	23.8	16.5	13.1	10.0	7.8	5.4	4.1	2.3	1.4	7,359
ME	82.2	60.9	44.9	35.0	27.3	23.0	18.2	14.3	9.8	6.7	4.4	2.2	3,818
MI	66.5	34.4	20.3	13.3	8.8	6.4	4.9	3.9	2.1	1.7	0.8	0.4	5,998
MN	96.6	80.0	62.8	49.2	35.5	26.4	21.0	16.2	9.1	5.5	2.6	1.3	1,783
MO	77.8	46.6	30.4	21.4	15.6	10.5	8.6	6.9	4.8	3.1	1.7	0.8	1,917
NC	71.7	49.5	34.3	23.5	16.6	12.4	9.8	7.6	4.9	3.2	1.9	1.4	12,277
ND	98.8	92.2	80.4	66.2	52.3	41.1	33.3	27.5	19.7	14.1	9.1	5.0	6,815
NE	98.2	88.6	75.4	61.4	48.5	39.3	29.5	23.3	13.0	8.7	4.9	2.5	4,566
NM	93.5	74.9	59.2	49.8	38.7	31.1	23.9	16.2	11.0	6.5	4.9	3.9	3,900
NV	89.9	73.5	55.7	47.8	40.0	32.9	28.3	25.6	20.1	14.5	11.8	6.5	1,728
OH	78.2	54.5	41.2	32.9	26.9	22.2	18.2	14.5	10.4	8.0	5.9	3.7	9,816
OK	71.3	43.4	25.3	17.6	14.7	9.8	7.9	5.9	4.5	1.0	1.0	1.0	910
PA	86.1	65.6	51.4	42.4	36.2	31.1	27.6	24.7	18.9	16.0	12.2	8.6	10,120
RI	77.1	45.7	29.5	21.6	16.8	12.9	9.2	6.6	4.9	3.5	2.0	1.7	2,372
SC	70.6	52.2	29.9	24.7	14.3	10.2	8.1	7.0	5.1	4.0	2.0	1.0	2,986
TN	83.9	58.5	43.5	32.9	23.9	18.1	15.6	13.8	10.3	8.4	6.8	4.9	3,696
VT	69.2	41.7	29.7	22.2	19.2	13.7	12.0	9.6	5.3	3.6	2.3	1.5	1,739
WI	82.3	58.2	41.8	27.8	19.4	13.0	9.5	6.7	4.3	3.2	1.8	0.8	3,297
WV	81.8	56.4	39.2	28.8	20.1	15.5	11.8	9.4	7.1	5.8	3.1	1.6	1,547
WY	88.8	62.2	45.3	32.5	23.3	15.7	12.7	10.1	7.1	5.3	4.2	2.6	2,020

[a] Estimates for Louisiana are not shown because of small sample size—only four houses.

sample than others. To counterbalance these unequal selection probabilities, appropriate sampling weights were calculated and used in analyzing the resulting screening measurements to assure unbiased statistical estimates (e.g., geometric mean, arithmetic mean).

Statewide estimates

The arithmetic mean (AM), geometric mean (GM), and geometric standard deviation (GSD) are shown in Table 3 for basement houses, nonbasement houses, and all houses combined. As expected, ^{222}Rn measurements taken in livable basements are, as a group, higher than measurements made on the first floor of nonbasement houses. This is clearly evident in each of the 30 states surveyed. Basement AMs ranged from 67 Bq m^{-3} (1.8 pCi L^{-1}) in Arizona and California to 348 Bq m^{-3} (9.4 pCi L^{-1}) in Iowa. Nonbasement AMs ranged from 19 Bq m^{-3} (0.5 pCi L^{-1}) in Louisiana to 204 Bq m^{-3} (5.5 pCi L^{-1}) in Iowa. On a statewide basis, Louisiana has the lowest and Iowa has the highest ^{222}Rn level. Overall comparisons among states can be misleading because of statewide differences in the percentages of houses with basements. For instance, Kansas and New Mexico are similar in that both have statewide AMs of 115 Bq m^{-3} (3.1 pCi L^{-1}). This similarity between the states is not evident when basement houses are compared or when nonbasement houses are compared—the AM in New Mexico is 49% higher than in Kansas for basement houses and 88% higher for nonbasement houses. The reason why both statewide estimates of the AM are the same is because Kansas has a higher portion of houses with livable basements—63.2% as compared to 5.4% in New Mexico.

The GMs for basement houses range from 33 Bq m^{-3} (0.9 pCi L^{-1}) in California to 241 Bq m^{-3} (6.5 pCi L^{-1}) in Iowa. For nonbasement houses, Louisiana has the lowest GM (11 Bq m^{-3}, or 0.3 pCi L^{-1}) and Iowa has the highest (134 Bq m^{-3}, or 3.6 pCi L^{-1}). Although not included in this paper, estimates of the median (i.e., 50th percentile) were derived for basement and nonbasement houses in each state. In each case, the estimates of the median and the GM were very similar—an indication that screening measurements may be

Table 5. Statewide distributions of first floor screening measurements taken in nonbasement houses.

State	Estimated percentage of houses with screening measurements >X Bq m^{-3} (pCi L^{-1}) for X equal to												Maximum value (Bq m^{-3})
	37 (1)	74 (2)	111 (3)	148 (4)	185 (5)	222 (6)	259 (7)	296 (8)	370 (10)	444 (12)	555 (15)	740 (20)	
AL	32.9	10.9	4.7	3.0	2.2	1.7	1.4	1.1	0.7	0.4	0.2	0.1	4,777
AZ	52.6	23.4	11.8	6.3	3.1	1.5	0.6	0.5	0.4	0.2	0.1	0.1	1,880
CA	30.6	10.1	3.5	2.1	1.8	1.2	0.8	0.5	0.2	0.2	0.2	0.1	1,010
GA	52.9	18.2	8.3	3.7	2.1	1.4	1.1	0.8	0.4	0.2	0.1	0.0	611
IA	89.4	72.2	60.3	51.0	40.6	31.9	27.1	18.6	12.0	8.8	4.8	2.7	1,358
ID	59.0	32.0	19.8	10.2	7.9	4.8	4.1	2.4	1.9	1.6	0.2	0.2	2,009
IN	61.4	36.5	23.7	17.3	12.8	8.7	6.0	4.4	2.8	1.6	1.0	0.7	1,691
KS	52.2	27.3	12.6	5.9	3.9	2.1	1.6	0.8	0.4	0.4	0.3	0.1	1,184
KY	41.1	17.7	10.8	8.0	6.2	4.2	3.6	3.2	1.7	0.8	0.6	0.1	936
LA	12.8	3.1	1.1	0.9	0.5	0.2	0.2	0.0	0.0	0.0	0.0	0.0	296
MA	46.8	26.5	17.3	14.9	10.2	9.4	8.6	7.9	6.3	3.1	2.4	1.6	888
ME	48.7	23.5	13.7	10.0	4.9	2.6	1.8	1.8	1.6	0.8	0.8	0.0	707
MI	28.1	12.5	6.7	4.1	2.5	1.0	0.7	0.7	0.4	0.3	0.3	0.0	670
MN	77.5	54.6	32.1	24.2	18.0	13.2	10.5	8.7	7.7	3.9	1.8	1.4	940
MO	50.6	27.4	14.4	9.2	5.0	3.2	2.2	2.0	1.3	0.9	0.4	0.4	1,388
NC	24.5	8.5	5.2	2.4	1.4	0.7	0.7	0.3	0.2	0.2	0.1	0.0	696
ND	78.5	58.4	40.5	27.8	18.7	14.9	10.3	8.1	5.1	3.2	0.8	0.0	611
NE	76.8	53.3	36.0	24.7	18.4	11.2	9.8	5.1	2.8	0.8	0.1	0.0	585
NM	82.2	52.3	32.0	20.6	14.2	9.3	6.7	5.1	3.8	2.0	1.2	0.7	3,230
NV	47.2	20.1	10.9	6.3	3.6	2.0	1.6	1.5	1.0	0.5	0.2	0.1	973
OH	60.2	38.9	23.6	16.1	12.7	8.4	7.1	6.3	3.6	2.8	1.1	0.2	844
OK	35.1	13.7	5.3	2.5	1.8	1.3	1.0	0.7	0.7	0.4	0.3	0.0	685
PA	61.7	44.4	29.5	25.0	21.4	15.9	12.8	9.6	6.4	5.1	3.2	2.0	1,584
RI	46.7	22.2	11.1	7.5	3.8	3.8	3.8	3.8	3.8	3.8	3.8	3.8	966
SC	27.3	8.3	2.9	1.6	1.1	0.9	0.5	0.4	0.2	0.2	0.2	0.2	1,709
TN	52.0	26.2	15.0	10.3	5.9	4.4	3.5	2.5	1.7	1.4	0.8	0.2	921
VT	33.9	16.9	9.9	6.6	4.9	3.1	2.1	1.8	1.1	0.8	0.8	0.0	577
WI	48.6	14.3	7.9	6.7	4.5	0.2	0.0	0.0	0.0	0.0	0.0	0.0	252
WV	43.5	18.0	8.4	4.7	3.2	2.4	1.6	1.3	0.9	0.7	0.5	0.2	3,038
WY	58.4	32.9	20.3	11.6	7.6	4.6	4.3	3.5	2.1	1.2	0.4	0.4	992

approximated by a lognormal distribution. The GSDs were about the same for basement and nonbasement houses. Across all 30 states, most GSDs fall between 2.5 and 3.5. In calculating the GM and GSD, all screening measurements ≤0.0 Bq m^{-3} were set equal to 2 Bq m^{-3} (0.05 pCi L^{-1}) prior to taking logarithms. There are more sophisticated ways of handling nonpositive readings (Gilbert 1987), but the low percentage found in these surveys did not warrant their use. A negative reading occurs whenever the observed number of counts is less than the background count established for the counting device.

Distributions of screening measurements of ^{222}Rn taken in basements are shown in Table 4. For each state, the estimated percentage of houses in the target population exceeding X is given for 12 values of X ranging from 37 Bq m^{-3} (1.0 pCi L^{-1}) to 740 Bq m^{-3} (20.0 pCi L^{-1}). The upper limit of 740 Bq m^{-3} equals or exceeds the 95th percentile in 26 of the 30 states.

The 12 percentages shown in Table 4 for a given state may be used to approximate percentiles of interest (e.g., 50th, 90th) or may be plotted on logarithmic probability paper as a visual check of the assumption that screening measurements are lognormally distributed (plotted points will lie on a straight line if the assumption is true). Table 4 may also be used to determine for the various states the expected percentages of houses exceeding a specific level. For example, more than 60% of the houses with livable basements in Iowa, Minnesota, North Dakota, and Nebraska exceed 111 Bq m^{-3} (3.0 pCi L^{-1}). Furthermore, the estimated percentages of houses with livable basements for the four states are high—ranging from 79% to 89% (Table 2).

Distributions of screening measurements taken on the first floor of nonbasement houses are given in Table 5 for the 30 states. For completeness, statewide distributions of screening measurements taken in both basement and nonbasement houses are presented in Table 6. Note that the format of Table 4 is maintained in constructing Tables 5 and 6. Consequently, the general comments in the previous paragraph apply equally to the distributions shown in Tables 5 and 6. A plot of the individual state percentages given in Table 6 under the column heading of 148 Bq m^{-3} is shown in Fig. 1.

The extent to which observed distributions depart from a lognormal distribution characterized by the parameters geometric mean and geometric standard deviation is discussed later in this paper using screening measurements from all 30 states.

30-state area estimates

Screening measurements from 43,054 houses tested in 30 state surveys, along with the corresponding sampling weights, were combined into one data set and

Table 6. Statewide distributions of screening measurements taken on the lowest livable level in all houses.

State	Estimated percentage of houses with screening measurements >X Bq m^{-3} (pCi L^{-1}) for X equal to												Maximum value (Bq m^{-3})
	37 (1)	74 (2)	111 (3)	148 (4)	185 (5)	222 (6)	259 (7)	296 (8)	370 (10)	444 (12)	555 (15)	740 (20)	
AL	39.6	17.4	9.6	6.4	4.1	3.0	2.5	2.0	1.3	0.7	0.5	0.3	6,660
AZ	51.9	23.5	12.0	6.5	3.3	1.7	0.8	0.6	0.4	0.3	0.2	0.1	1,880
CA	31.6	10.5	3.8	2.4	2.0	1.4	0.9	0.6	0.3	0.3	0.3	0.1	1,077
GA	60.6	26.8	14.1	7.5	5.0	3.6	2.2	1.7	1.0	0.7	0.3	0.0	692
IA	96.1	88.0	79.7	71.0	62.9	56.2	48.9	40.9	31.2	23.0	14.4	7.5	4,814
ID	70.7	47.2	31.3	19.3	15.2	12.7	10.7	9.2	7.7	6.0	2.7	1.8	2,194
IN	73.7	51.9	37.9	28.5	21.2	15.7	12.5	10.1	6.8	5.0	3.5	1.5	2,657
KS	74.2	49.9	33.9	22.5	15.7	11.7	8.5	6.1	3.5	2.5	1.8	0.7	1,776
KY	56.0	32.2	21.0	17.1	13.8	11.0	9.4	8.0	4.6	3.3	2.2	1.5	2,424
LA	12.3	3.1	1.1	0.8	0.4	0.2	0.2	0.0	0.0	0.0	0.0	0.0	296
MA	73.5	46.6	31.6	22.7	15.8	12.6	9.8	7.8	5.4	3.8	2.2	1.3	7,359
ME	75.6	53.2	38.9	29.9	22.8	19.0	15.0	11.7	8.2	5.6	3.8	1.9	3,818
MI	59.6	30.6	17.8	11.7	7.7	5.5	4.1	3.3	1.8	1.4	0.7	0.4	5,998
MN	93.7	76.2	58.2	45.4	32.8	24.4	19.4	15.0	8.8	5.3	2.5	1.4	1,783
MO	68.2	39.8	24.7	17.0	11.8	7.9	6.3	5.2	3.6	2.4	1.2	0.7	1,917
NC	33.5	16.6	11.1	6.7	4.4	3.1	2.5	1.8	1.2	0.8	0.4	0.3	12,277
ND	95.9	87.4	74.7	60.7	47.5	37.4	30.0	24.7	17.6	12.6	8.0	4.3	6,815
NE	93.6	81.1	66.8	53.5	41.9	33.2	25.0	19.2	10.7	7.0	3.8	1.9	4,566
NM	82.4	52.9	33.1	21.9	15.2	10.3	7.5	5.6	4.1	2.2	1.4	0.8	3,900
NV	52.3	25.7	15.2	10.2	7.1	5.0	4.2	3.9	3.0	1.9	1.4	0.8	1,728
OH	73.8	50.8	37.0	29.0	23.4	18.8	15.5	12.6	8.7	6.8	4.7	2.8	9,816
OK	36.1	14.8	6.1	3.3	2.5	1.7	1.4	1.0	0.9	0.4	0.3	0.0	910
PA	83.5	63.3	49.0	40.5	34.6	29.5	26.0	23.1	17.6	14.9	11.2	7.9	10,120
RI	74.8	44.0	28.1	20.6	15.8	12.2	8.8	6.4	4.8	3.5	2.2	1.9	2,372
SC	31.2	12.2	6.4	3.7	2.3	1.7	1.3	1.1	0.7	0.6	0.4	0.3	2,986
TN	59.7	33.9	21.7	15.8	10.4	7.8	6.4	5.2	3.8	3.1	2.2	1.3	3,696
VT	55.6	31.7	21.8	15.9	13.5	9.5	8.0	6.5	3.6	2.5	1.7	0.9	1,739
WI	79.8	55.4	39.4	26.6	18.6	12.1	8.9	6.4	4.2	3.2	1.8	0.8	3,297
WV	61.2	35.5	22.5	15.7	10.9	8.4	6.3	5.1	3.9	3.2	1.7	0.8	3,038
WY	79.0	53.3	37.8	26.2	18.7	12.3	10.2	8.0	5.6	4.1	3.0	1.8	2,020

analyzed. The results are summarized in Table 7. These results apply to an estimated population of 22 million houses located over a 30-state area that had a positive probability of being selected for testing.

The AM, GM, and GSD are shown in the upper portion of Table 7 for basement houses, for nonbasement houses, and for all houses. For all houses, the AM is 124 Bq m^{-3} (3.4 pCi L^{-1}), the GM is 54 Bq m^{-3} (1.5 pCi L^{-1}), and the GSD is 3.9. As noted previously for individual state surveys, the AM and GM are much higher in basement houses than in nonbasement houses. Basement houses exhibit a GSD of 2.9 as compared to 3.6 for nonbasement houses.

The estimated percentages of houses with screening measurements exceeding selected concentration values are shown in the lower portion of Table 7 for basement

Table 7. Summary results by house type for 30 state surveys.

House type	Arithmetic mean (Bq m^{-3})	Geometric mean (Bq m^{-3})	Geometric standard deviation	Number of houses tested	Number of houses in target population (in 1000s)	Percentage of houses with a basement
Basement	185 ± 3.3[a]	99 ± 0.4[a]	2.9	20,768	11,375	
Nonbasement	54 ± 0.8	27 ± 0.6	3.6	20,880	9,536	
All houses	124 ± 1.9	54 ± 0.4	3.9	43,054[b]	21,551	54.4

House type	Estimated percentage of houses with screening measurements >X Bq m^{-3} (pCi L^{-1}) for X equal to											
	37 (1)	74 (2)	111 (3)	148 (4)	185 (5)	222 (6)	259 (7)	296 (8)	370 (10)	444 (12)	555 (15)	740 (20)
Basement	81.8	58.7	43.9	34.0	26.5	21.4	17.6	14.6	10.3	7.9	5.3	3.3
Nonbasement	41.7	19.6	10.7	7.0	5.0	3.4	2.6	2.0	1.3	0.9	0.5	0.2
All houses	63.0	40.5	28.4	21.4	16.5	13.0	10.6	8.7	6.1	4.6	3.1	1.9

[a] Standard error of the parameter estimate.
[b] Includes houses for which basement/nonbasement classification was unknown.

Table 8. State survey results for nonbasement houses contrasted with published national statistics.

Parameter	30 state surveys (nonbasement houses)	Nero et al. (1986)	Cohen (1986)	Alter and Oswald[a] (1987)
Arithmetic mean				
Bq m^{-3}	54	55	54	160
pCi L^{-1}	1.5	1.5	1.5	4.3
Geometric mean				
Bq m^{-3}	27	33	38	63
pCi L^{-1}	0.7	0.9	1.0	1.7
Geometric standard deviation	3.6	2.8	2.4	3.5

[a] Estimates do not include data from six states that had more than 1000 measurements.

houses, for nonbasement houses, and for all houses. For consistency, the 12 selected concentration values are the same as those used to characterize the distributions within individual states. A more detailed description of the distributions of screening measurements for the 30-state area is given in the next section.

As noted earlier, ^{222}Rn concentrations at the national level were reported by Nero et al. (1986), Cohen (1986), and Alter and Oswald (1987). There are notable differences in these studies and the state surveys. First, these studies used a self-selected sample of households as opposed to a probability-based sample. Second, these studies used detectors located in the usual or main living area rather than in the lowest living area. Third, these studies used test periods up to 1 y as contrasted to a 2-d exposure period. Because of these fundamental differences, close agreement between published national level concentrations of indoor ^{222}Rn and the findings of the 30 state surveys is not necessarily anticipated.

The AM, GM, and GSD provided by the three referenced studies are shown in Table 8 along with corresponding results for nonbasement houses in the state surveys. Results for nonbasement houses are shown because the referenced studies used measurements taken primarily on the first floor. This provides a more meaningful basis for making comparisons. The results reported by Nero et al. (1986) and Cohen (1986) are similar and, for the most part, agree with the findings in the state surveys even though the latter are based on 2-d screening measurements taken during the winter season under closed-house conditions. The national level concentrations reported by Alter and Oswald (1987) are significantly higher than the other published results, probably because of the positive bias expected in a data base containing multiple test results from homeowners who elect to have their houses tested: Many of these homes may be located in high-risk areas.

Adequacy of lognormal distribution

There are many who believe that the distribution of indoor ^{222}Rn concentrations in randomly selected houses can be adequately approximated by a lognormal distribution with parameters geometric mean and geometric standard deviation, and there is evidence to support this belief (Cohen 1986; Nero et al. 1986; Alter and Oswald 1987; Fjeld et al. 1990). There are others who feel that a lognormal distribution tends to underestimate the actual percentages in the upper tail, namely, houses with very high concentrations (Goble and Socolow 1989; Levy and Small 1990).

The extent to which the observed distribution of screening measurements can be approximated by a lognormal distribution is examined using data aggregated over all 30 states. First, a table of relative frequencies is constructed from observed screening measurements. Because of the large number of houses tested over the 30-state area, the table contains 33 intervals—the first interval includes concentrations $\leq$37 Bq m^{-3} (1.0 pCi L^{-1}) and the last is an open-ended interval that includes concentrations >2960 Bq m^{-3} (80.0 pCi L^{-1}).

Next, the relative frequency expected under a lognormal distribution is calculated for each of the 33 intervals using the geometric mean and geometric standard deviation estimated from the observed screening measurements. Judgment of how well the lognormal distribution approximates the observed distribution is then made by simply comparing the two relative frequency distributions on an interval-by-interval basis. This approach for assessing the appropriateness of a lognormal distribution is applied to 1) 20,768 screening measurements taken in basement houses, 2) 20,880 screening measurements taken in nonbasement houses, and 3) 43,054 screening measurements taken in both basement and nonbasement houses. The results are shown in Table 9.

Normal probability plots (using weighted cumulative frequencies) for basement houses, for nonbasement houses, and for all houses in the 30-state area are shown in Fig. 2. The solid lines represent the cumulative percent expected under the assumption of a lognormal distribution. Estimates of the parameters GM and GSD used in constructing these lines are given in Table 7. In each case, the plotted points fall on or near the solid line.

It is evident from Table 9 and Fig. 2 that screening measurements in basement houses, in nonbasement houses, and in all houses as a group can be adequately approximated by a lognormal distribution. It should be noted that at a concentration of 1480 Bq m^{-3} (40.0 pCi L^{-1}), the lognormal distribution begins to underestimate the observed distribution of basement measurements (Table 9). The magnitude of the underestimate is, however, quite small. For example, 0.21% of the observed basement readings are greater than 2960 Bq m^{-3} (80.0 pCi L^{-1}), as compared to 0.08% estimated from the lognormal distribution. On the other hand, at higher concentrations, the lognormal distribution overestimates by a very small margin the observed distribution of measurements in nonbasement houses (Table 9).

Table 9. Comparison, by house type, of the observed relative frequency distribution of screening measurements over a 30-state area with relative frequencies expected under a lognormal distribution.

Interval (Bq m^{-3})	Basement houses		Nonbasement houses		All houses	
	Observed	Lognormal	Observed	Lognormal	Observed	Lognormal
≤37	18.19	18.01	58.29	59.67	36.98	39.06
38–74	23.12	21.34	22.09	18.60	22.56	20.25
75–111	14.77	14.91	8.91	8.06	12.03	11.09
112–148	9.97	10.34	3.68	4.29	7.02	6.90
149–185	7.41	7.37	2.04	2.57	4.92	4.67
186–222	5.17	5.41	1.60	1.67	3.50	3.32
223–259	3.75	4.07	0.78	1.14	2.36	2.47
260–296	3.04	3.13	0.63	0.81	1.91	1.90
297–333	2.34	2.46	0.41	0.60	1.45	1.48
334–370	1.94	1.95	0.28	0.45	1.17	1.19
371–407	1.40	1.58	0.22	0.35	0.84	0.97
408–444	1.00	1.29	0.20	0.28	0.64	0.80
445–481	1.07	1.06	0.14	0.22	0.63	0.67
482–518	0.86	0.89	0.13	0.18	0.52	0.56
519–555	0.64	0.74	0.09	0.15	0.39	0.48
556–592	0.59	0.63	0.06	0.12	0.34	0.41
593–629	0.46	0.54	0.05	0.10	0.27	0.36
630–666	0.30	0.46	0.04	0.09	0.18	0.31
667–703	0.34	0.40	0.05	0.07	0.21	0.27
704–740	0.38	0.35	0.04	0.06	0.22	0.24
741–777	0.29	0.30	0.05	0.05	0.17	0.21
778–814	0.20	0.26	0.01	0.05	0.12	0.19
815–851	0.20	0.23	0.04	0.04	0.12	0.17
852–888	0.22	0.20	0.02	0.04	0.12	0.15
889–925	0.18	0.18	0.01	0.03	0.10	0.14
926–1110	0.55	0.65	0.05	0.11	0.32	0.51
1111–1295	0.37	0.39	0.02	0.06	0.20	0.33
1296–1480	0.21	0.24	0.02	0.04	0.13	0.22
1481–1850	0.42	0.27	0.03	0.05	0.23	0.27
1851–2220	0.15	0.14	0.00	0.00	0.08	0.15
2221–2590	0.16	0.07	0.00	0.00	0.08	0.08
2591–2960	0.11	0.04	0.00	0.03	0.06	0.06
>2960	0.21	0.08	0.01	0.01	0.12	0.15

CONCLUSIONS

In every state, basement levels of ^{222}Rn were higher than first floor levels in nonbasement houses. Overall, the arithmetic mean of basement screening measurements was 185 Bq m^{-3} (5 pCi L^{-1}) as compared to 54 Bq m^{-3} (1.5 pCi L^{-1}) for first floor measurements in nonbasement houses. Levels of ^{222}Rn varied widely from one state to another: the arithmetic mean of 2-d measurements ranged from 19 Bq m^{-3} (0.5 pCi L^{-1}) in Louisiana to 329 Bq m^{-3} (8.9 pCi L^{-1}) in Iowa. Elevated screening results or concentrations were found in all states: 26 of the 30 states had at least one 2-d test result greater than 1700 Bq m^{-3} (46 pCi L^{-1}), and 12 states had at least one test result greater than 3700 Bq m^{-3} (100 pCi L^{-1}).

Basement screening results in the 30-state area can be adequately approximated by a lognormal distribution with a geometric mean (GM) = 99 Bq m^{-3} (2.7 pCi L^{-1}) and a geometric standard deviation (GSD) = 2.9. Similarly, first floor results tended to follow a

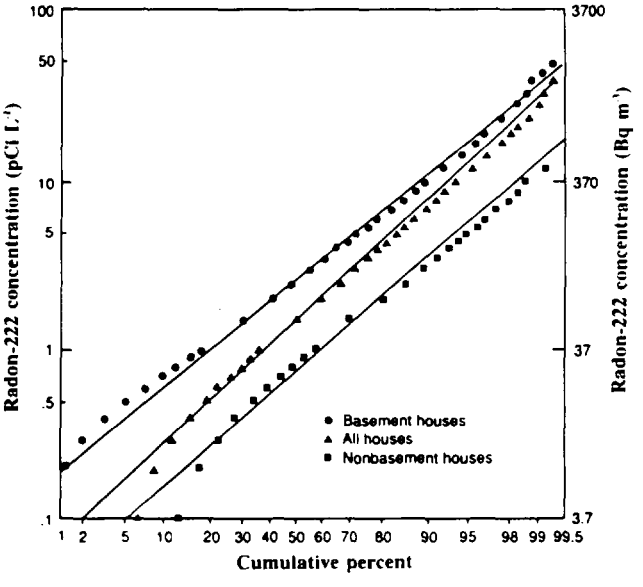

Fig. 2. Normal probability plots for houses in 30-state area.

lognormal distribution, with a GM = 27 Bq m^{-3} (0.7 pCi L^{-1}) and a GSD = 3.6. This distribution of first floor measurements compares favorably with the nationwide distribution reported by Nero et al. (1986), GM = 33 Bq m^{-3} (0.9 pCi L^{-1}) and GSD = 2.8, and the nationwide distribution reported by Cohen (1986), GM = 38 Bq m^{-3} (1.0 pCi L^{-1}) and GSD = 2.4. There is some evidence that, at high concentrations, a lognormal distribution underestimates, by a small margin, the observed distribution of basement measurements and overestimates (again by a small margin) the observed distribution of first floor measurements.

A wintertime screening measurement taken in the lowest livable level is an indicator of the maximum exposure to ^{222}Rn; it is not a direct measure of average concentration over extended periods. The 30 state surveys were not designed to characterize long-term (e.g., annual) levels of ^{222}Rn to which occupants are exposed. Thus, the results shown in this paper should not be used directly in making risk assessments.

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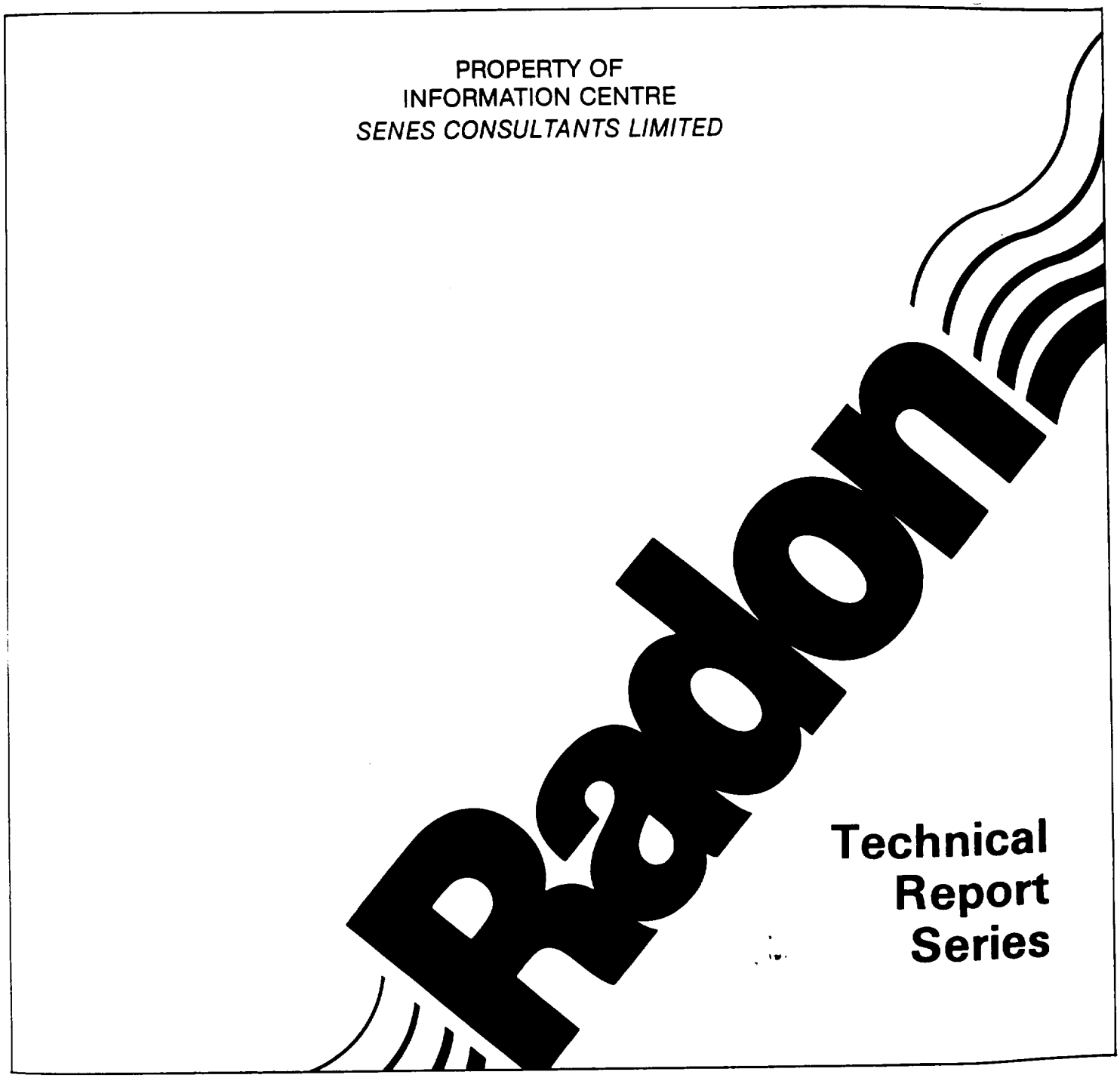

Technical Report Series

III. FACTORS AFFECTING INDOOR CONCENTRATIONS

The concentration of radon in homes and other buildings varies substantially from one area to another, from one structure to another, and even within the same structure. Likewise, in the same building there is often a substantial variation with time on various temporal scales, i.e., season to season, week to week, and on a daily or hourly basis. Understanding the nature and origin of such variability is important as a basis for evaluating the scope of the radon problem (and indeed simply for interpreting monitoring data) and for formulating effective strategies for control. We first examine the dependence of concentration on time, and to a lesser extent on location, within the same structure. We then examine the sources and transport of radon, as a basis for understanding where high concentrations are likely to occur and for designing effective techniques for reducing the rate of entry of radon into the indoor atmosphere.

A. Temporal and Spatial Dependence

The previous section reviewed U.S. monitoring data, indicating not only the variability of indoor concentrations from one area to another, but also the critical dependence of concentration measurements on the time period and location of monitoring. The question of monitoring techniques and protocols is the subject of a later section, but we here review briefly some of the primary information on temporal and spatial dependencies, both to understand better the interpretation of monitoring data and as a further indication of the importance of differences in sources and transport processes in determining the indoor concentration.

Numerous measurements of radon in dwellings have indicated the substantial variation with time of ^{222}Rn and its decay products. This variability occurs with time of day, weather conditions, or season. Without ascribing any cause to this variability - which is certainly related to environmental or operational parameters affecting radon entry or removal (or behavior of the decay products) - we here indicate the nature of time variability by example.

One of the earlier examples of short-term variability was obtained using continuous ^{222}Rn monitors in a New Jersey house, measuring basement and upper floor concentrations over a period of weeks (Spitz et al. 1980). A significant diurnal dependence was observed, consistent with later measurements, which however sometimes showed a stronger difference between maxima and minima.

Subsequent work has examined the dependence of concentrations on various factors, directly measuring not only the concentration, but also source parameters and environmental factors affecting radon entry and ventilation rates (Nazaroff et al. 1985). An example of such real-time measurements is shown in Figure 9. Detailed examination of such correlations is important for understanding radon entry and removal, as discussed later in Section B.

More recent studies of radon entry have provided further information on the variability of indoor concentrations over different time scales. For example, Figures 10 and 11 show data from one of 14 New Jersey houses in which measurements examining the effectiveness of control techniques were performed over a period of approximately one year. Figure 10 shows data in which short-term variations can be seen, while Figure 11 shows results from the same house after averaging over 3-day periods. In either case, substantial variability is evident.

It is also known that substantial seasonal variability occurs, as is often seen between winter and year-long measurements. For example, Table 4 shows a comparison of winter and annual-average results from four different studies from Nero et al. (1986), giving the average and lognormal parameters. In these studies, the average of measured annual-average concentrations

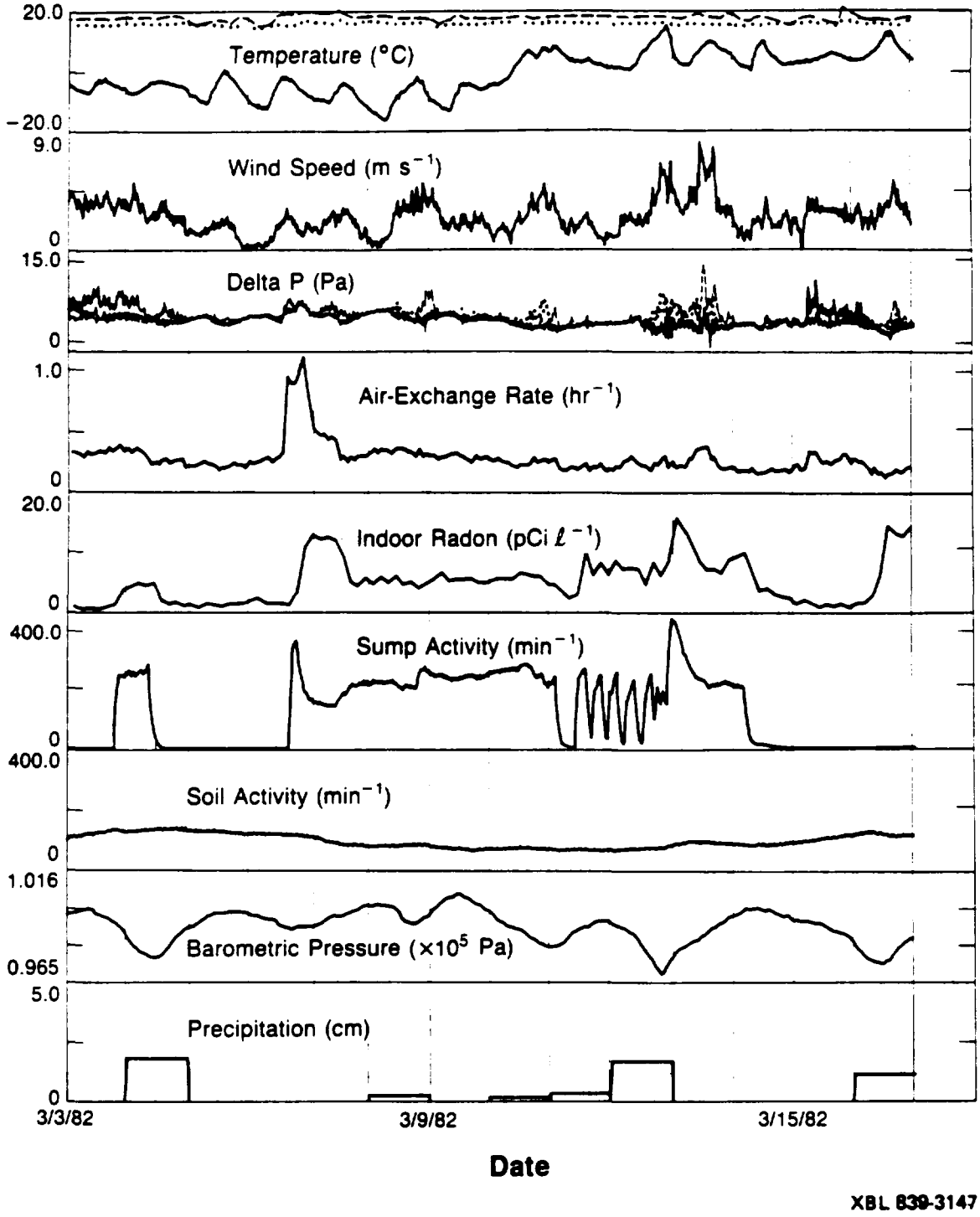

Figure 9. Variability of ^{222}Rn concentration, ventilation rate, and environmental parameters in a house in a basement.

These data were accumulated over a two-week period during a several-month experiment examining the dependence of radon entry on environmental factors (Nazaroff et al. 1985).

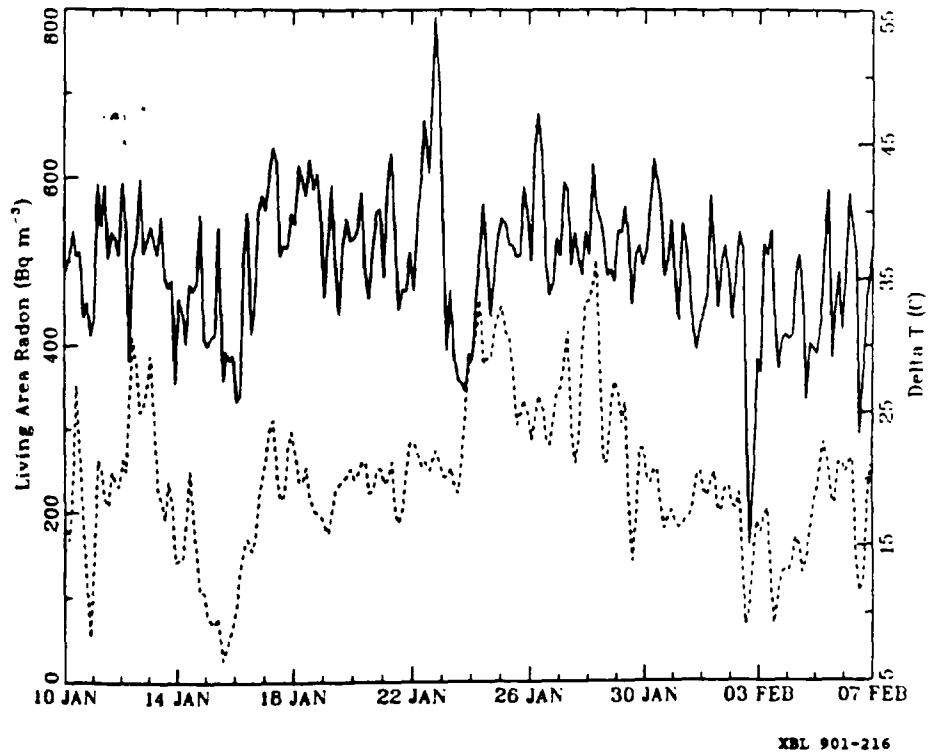

Figure 10. Radon concentration in the living space as a function of time.

Data are from a house studied in New Jersey (cf. Revzan 1989). Also shown (dashed line) is the temperature difference between indoors and outdoors.

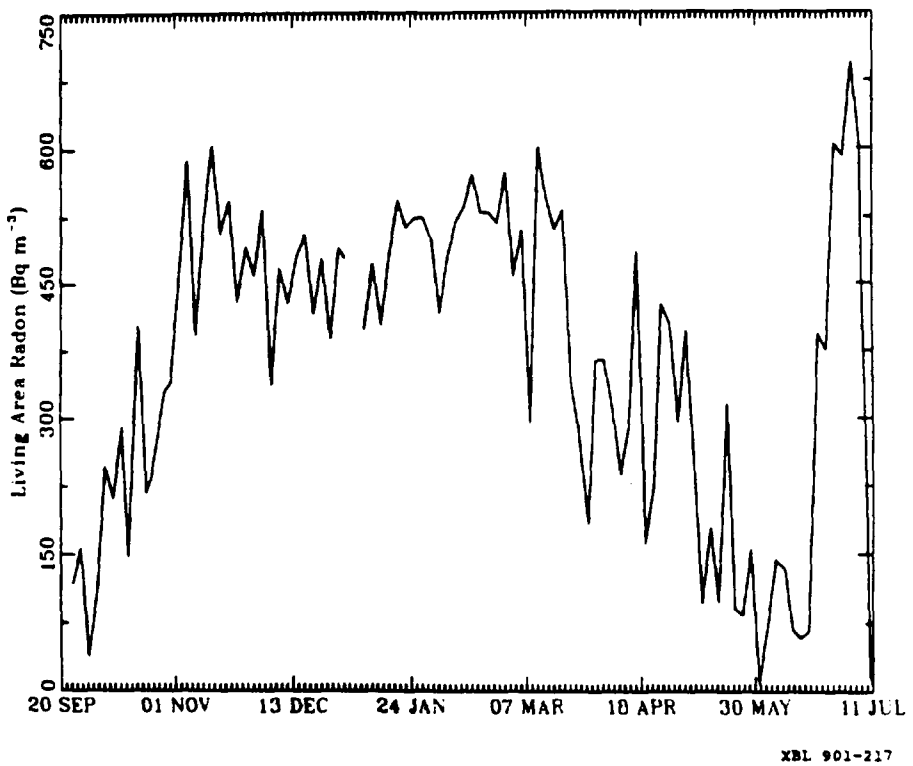

Figure 11. Radon concentration in the living space as a function of time, averaged over three-day periods.

Data are from the same house as in Figure 10.

is less than the average winter values (although the difference is modest in the case of Maine). If we calculate the ratio of annual-average concentrations to winter concentrations, this is found to average 0.72. This difference has important implications for the potential interpretation of the large-scale data sets acquired primarily since 1985, discussed previously. Equally significant, in this case for assessing the prevalence of higher-than-average concentrations, is that the width of the annual-average distribution is also smaller, as given by the GSD. Assuming the GSD was the same, and that the entire distribution was merely scaled in proportion to the average in going from winter to annual-average, could thus overestimate the fraction of homes having concentrations above levels of concern such as 150 Bq/m^3 and 740 Bq/m^3. In a more recent study in Colorado, Borak et al. (1989) found - using etched-track detectors - the ratio of measured annual concentration to the average of summer and winter measurement results to have a mean of 1.0 ± 0.3. On the other hand, the GSDs for winter and summer were similar to that for year-long measurements.

The New York State survey (Hartwell et al. 1987) gives an indication of the effects of seasonal differences, based on a statistical sample. This effort gave an average of 41 Bq/m^3 (1.1 pCi/l) for year-long measurements in the living space, in contrast to 52 Bq/m^3 (1.4 pCi/l) for 2-month winter measurements. This yields a ratio of 0.79, in the range from Nero et al. (1986). Percentages greater than 148 Bq/m^3 (4 pCi/l) were 4.2% and 5.0% for year-long and winter measurements, respectively, showing only a slight difference. The results of Borak et al. (1989) show a greater difference, i.e., a winter to annual ratio of approximately 0.65 and percentages greater than 150 Bq/m^3 of 50% for winter and 20% for annual measurements.

Turning to spatial differences, the New York State study found a substantial difference between basement and living-space concentrations, with basement values averaging 100 Bq/m^3 (2.7 pCi/l), based on year-long measurements, approximately 2.5 times the living-space average. This factor is generally consistent with other information. For example, Alter and Oswald (1987) found a ratio of 2.0, as noted above, although this is generally for shorter-term measurements. And Cohen (1989) finds an average in basements of 249 Bq/m^3 (6.7 pCi/l), a factor of 2.0 times the average from living-space measurements. The New York ratio may be larger than these because it is a comparison of year-long measurements and because, whereas the living-space value tends to change substantially between winter and summer, basement values are thought to be more stable. Thus, for example, the ratio basement (year-long) to living-space (winter) measurements is 1.9, quite close to the other results cited. Finally, there is significant evidence that, for houses with higher than average concentrations, basement concentrations may exceed those in living areas by more than a factor of 2, an example being the factor of 5 differential for higher-concentration houses, found in NJSDH 1989.

B. Sources and Transport Processes

Radon arises from trace concentrations of radium in the earth's crust, and indoor concentrations depend on access of this radon to building interiors. Radon can enter directly from soil or rock, via utilities such as water (and, in principle, natural gas) that carry radon, or from crustal materials that are incorporated into the building structure in the form of concrete, rock, and brick. The relative importance of these pathways depends on the circumstances, but it has become clear that the first - direct ingress from the soil - ordinarily dominates the higher indoor concentrations that have been observed in homes.

Indications of this arose in early investigations of radon in U.S. houses, when it was found that measurements of radon emanating from structural materials could not account for observed indoor concentrations, based on estimates of the air exchange rate (George and Breslin 1980). Moreover, practical experience in reducing concentrations in the Canadian mining communities made it clear that the major entry route was through the house understructure, at least in the houses in which remedial measures were attempted (Scott 1988).

ANDREWS OFFICE PRODUCTS CAPITOL HEIGHTS, MD (K)

NCRP REPORT No. 97

MEASUREMENT OF RADON AND RADON DAUGHTERS IN AIR

Recommendations of the
NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS

Issued November 15, 1988

National Council on Radiation Protection and Measurements
7910 WOODMONT AVENUE / BETHESDA, MD 20814

possible, and this poses a difficult problem for estimating exposure. It is emphasized that an attempt should be made to obtain the best data which can be used in estimating annual averages.

Identification of areas with a high radon risk can be carried out with short-term measurements of either radon or the daughter products. These measurements are usually performed under conditions designed to maximize the result, for example, sampling after the house has been closed for some time and sampling in the basement rather than the living quarters.

Scientific studies may make use of any of the techniques described, but tend to lean on instantaneous or continuous recording devices. Additionally, such studies frequently are designed to seek out sources or pathways and aim at sampling limited areas within a house.

Measurements made while conducting remedial action are generally short-term or grab sampling to identify a persistent source. Once remediation is completed, long-term measurements are necessary to demonstrate that the annual average exposure has been reduced to desired levels. In some instances, where elevated and unevenly distributed ^{226}Ra in soil is the source, flux measurements made at many spots on walls and floors can help to identify anomalously high entry points for radon.

13.2 Short-Term Versus Long-Term Measurements

The choice of method for exposure assessment is usually dictated by the instrumentation and effort available. The simplest methods are short-term or grab sampling methods and require repeat measurements for the estimation of annual average exposure.

The simplest on-site method is the single alpha count of a filtered air sample by the Kusnetz or Rolle method. This provides a rapid measurement of the radon daughter working level. It is more prudent to alpha count this same sample three times rather than only once since this allows evaluation not only of the WL but also the concentrations of the specific daughters nuclides. The Thomas method or the Scott MRK, (Scott 1981) method provides these data and the Scott method utilizes the second or third count interval to directly compute working level.

Alpha counting a filtered air sample provides immediate results and is useful in screening. Low background, high-efficiency alpha counters are available so that even outdoor radon daughter concentrations may be measured this way. Samples should be recounted after from 3 hours

to one day to ensure that thoron daughter activity is not an interference. If it is possible to utilize an alpha spectrometer for the measurement, the detection limit is usually improved because of the ability to determine ^{218}Po and ^{214}Po separately, and to distinguish these immediately from any thoron daughters.

If a WL meter is available, this provides a spot check of the WL present at one point in time but repeated measurements are necessary to provide an estimate of annual exposure.

Spot radon measurements are possible by collecting air samples. The collecting vessel is usually a counting flask lined with zinc sulfide alpha phosphor. This can be placed directly on a photomultiplier tube for measurement. The containers, often called Lucas flasks, vary in volume from 0.1 liter to many liters. Counting can commence as soon as radioactive equilibrium between radon and the daughters is established (about three hours). These flasks are equipped with either one or two entrance ports for filling. The single-port flask is less desirable since it must generally be evacuated and taken to the site and opened for filling. This relies on the integrity of the vacuum seal. The two-port flask allows air to be drawn through the flask by a small pump at the site, and filling is complete and unambiguous. Counting of either type is usually performed upon return to a central site or laboratory.

The radon grab sample provides immediate results and is useful for screening purposes but has the same drawbacks as any instantaneous sample for estimating annual exposure and repeat measurements are necessary.

Longer-term radon measurements are possible with charcoal samplers (two to seven days) and this technique is easy to implement. Samplers can be deployed by individual home owners and samplers can be sent out and returned by mail to the central site for gamma counting of the radon daughters. These are in equilibrium with the adsorbed radon on the charcoal within a few hours. Repeat measurements are necessary for determining an annual average radon exposure.

Harley and Terilli (1988) report that, based on two different homes studied for 18 months, two one-week measurements, taken one in summer and one in winter, and averaged, will yield within $\pm 50\%$ of the actual measured annual average ^{222}Rn concentration.

Other longer-term radon monitoring measurements, with deployment for periods of up to a few months, can be performed with TLD based monitors and for up to a full year with solid state nuclear track detectors.

A strategy for rapid measurement of the radon level in a home is sometimes necessary. This usually is required for the buying or selling

of homes but other situations such as assessment of a workplace or public building can arise. In this case a valid measurement of radon concentration over a short time interval with charcoal, working level monitor or even grab radon samples taken in a living or working area may suffice to provide some guidance. This measurement should not be confused with measurements for actual human exposure which must be conducted on a longer term basis.

A short term (say two to four day) sample taken in the basement of an average home in the winter might yield a value of perhaps 200 Bq m^{-3} (4 pCi l^{-1}). A decision that exposure of the occupants is in the range necessary for remedial action cannot be made from this measurement. First floor and second floor radon concentrations generally average 2 to 5 times lower than basement concentrations depending upon season and other variables related to the specific dwelling such as type of heating. Thus, 200 Bq m^{-3} in the basement most likely indicates the typical value of 40 to 100 Bq m^{-3} in the upstairs living space.

The single rapid measurement provides little useful information concerning compliance with any established guidelines for exposure of individuals within a dwelling (and thus the need for remediation). A short term measurement can provide information that a home has unusually high radon concentrations. Radon concentrations in excess of 2000 Bq m^{-3} regardless of season or location within the dwelling will assure that some radon reduction technique (NCRP, 1988b) is necessary in the dwelling to conform with guidelines (NCRP, 1984a).

Valid exposure measurements can only be obtained with long term follow-up integrating measurements or several measurements of a week's duration taken during a minimum of two seasons per year. The measurements for exposure must be conducted in the living space within the home.

13.3 Reporting of Data

The reporting of data should reflect the sampling intervals involved. The best estimate of exposure on an annual basis is with long term monitoring and the use of time weighting to include exposure out of doors and in buildings other than the home. Time weighted average exposures are calculated as follows:

$$\text{ANNUAL EXPOSURE (WLM)} = (1/170)\,[\,(\text{WL})_1(\text{Hours})_1 + \ldots + (\text{WL})_n(\text{Hours})_n] \qquad (13.1)$$

OFF-SITE THORIUM REMOVAL PROGRAM

UNINCORPORATED RESIDENTAL NEIGHBORHOODS

WEST CHICAGO, ILLINOIS

August 23, 1989

UNINCORPORATED WEST CHICAGO AREA, OFF-SITE THORIUM REMOVAL

Introduction

Thorium-containing materials have been identified at various locations in unincorporated areas near the City of West Chicago. The thorium bearing deposits are small in area and do not pose a health threat, but are both a source of adverse publicity and concern to the residents.

In recognition of this situation, Kerr-McGee Chemical Corporation is undertaking a voluntary program to remove certain thorium containing soil deposits from these unincorporated areas. Kerr-McGee is doing this in the spirit of cooperation and public interest, as Kerr-McGee does not have any legal responsibility to do so. The program elements are described below.

UNINCORPORATED RESIDENTIAL NEIGHBORHOOD THORIUM REMOVAL PROGRAM

The unincorporated area residential neighborhood program involves identifying the material to be removed, excavating and removing the material, restoring disturbed areas, and returning excavated materials to KMCC's licensed West Chicago facility. The removal program will exclude soils adjacent to and within waterways and below the existing water table. Excavated materials returned to the facility will be managed with similar materials previously removed from City of West Chicago properties.

A. Identifying Thorium-bearing Locations

1. Authorization: Oral permission to survey a property will be obtained by Kerr-McGee. The survey will be conducted by a trained Kerr-McGee survey team equipped with appropriate gamma field survey instruments.

2. Criterion: Any area on a property determined to have an external gamma radiation level that exceeds 35 micro-R per hour at one meter above the surface will be marked for removal subject to permission of the property owner.

3. Marking Areas: Areas identified as exceeding the removal criterion will be noted on sketches or maps of the property.

B. Removing Identified Materials

1. Excavating Thorium-Containing Material: Kerr-McGee will, in a careful and expeditious manner, remove identified materials using hand equipment and/or small machinery as appropriate to minimize property disturbances.

Excavation on a property will be started only after (i) Kerr-McGee has obtained the property owner's written permission on the form shown in Appendix 1, (ii) the existing physical conditions have been documented with photographs, and (iii) the homeownwer has applied for and received the appropriate county excavation/fill permit authorizations.

2. Excavated Material: Kerr-McGee will place the excavated material in appropriate containers, such as drums, metal boxes or trucks and cover the tops. Kerr-McGee will promptly transport the material to an unincorporated parcel of the Kerr-McGee West Chicago facility for management in accordance with USNRC license conditions.

3. Resurveying Excavated Areas: Kerr-McGee will resurvey the area excavated using gamma survey instrumentation to assure that the material has been removed to the extent practicable.

4. Exceptions: If an area marked for removal includes a structure, Kerr-McGee and the property owner will consult and agree on a course of action in consideration of the physical conditions and potential for exposure to the materials.

C. Restoring Excavated Areas

1. Refilling Excavated Areas: After areas have been excavated, Kerr-McGee will, without unnecessary delay, refill with clean soil the areas from which the thorium-containing material has been removed and re-establish previous contours. Appropriate equipment will be used so as to keep disturbances to a minimum. Any plants, shrubs or structures that were removed to allow excavation will be returned (or replaced as appropriate) to locations documented in the photos (large older plants and shrubs may have to be replaced with small younger plants and shrubs).

2. Resurveying After Restoration: Kerr-McGee will resurvey the restored areas to document the external gamma levels.

D. Managing Excavated Material

1. Hauling to the Facility: Kerr-McGee will haul excavated materials to the facility site. The vehicles will be unloaded in an unincorporated area inside the licensed facility site boundary.

2. Handling at the Facility: Kerr-McGee may relocate the materials brought to the facility site within the unincorporated area. Containers to be reused will be emptied and returned to the facility site perimeter.

3. Stockpiling at the Facility: Excavated material relocated by Kerr-McGee to the facility site will be stockpiled separately from other materials on the site. Documentation will be maintained showing origin of the excavated material, quantities received, date of receipt, and level of radiation.

4. Future Responsibility: After material excavated under the unincorporated residential neighborhood thorium removal program has been unloaded at the facility site, the property owner from whose property the material originated will not have any further responsibility for the care and management of that material.

E. Personnel Requirements and Equipment

1. Survey Team: The survey team will consist of trained personnel provided by Kerr-McGee. The team will be equipped with a gamma survey instrument, large scale maps or sketches, and other items as required or appropriate for identifying locations with thorium-containing materials greater than 35 micro-R per hour at one meter.

2. Excavation Team: Kerr-McGee will provide an appropriately-manned excavation team as dictated by conditions involved for a specific location. The team will include a trained radiation surveyor to guide the removal of the material and its placement in containers and to assure adherence to radiation safety practices. Excavation equipment will consist of shovels, picks and other hand tools and where feasible, small mechanical diggers/loaders.

3. Hauling: The excavated material will be hauled by truck to the Kerr-McGee West Chicago facility.

4. Facility Handling Team: Trained Kerr-McGee site personnel will handle the material delivered to the facility.

5. Restoration Team: An appropriately manned restoration team and necessary restoration equipment will be provided by Kerr-McGee.

F. TIMETABLE AND MANNING

The unincorporated neighborhood thorium-containing material removal program is effective the date of this document. Removal work at specific properties will begin at a time agreed to by Kerr-McGee and the property owner following receipt of the property owner's oral authorization to survey and the identification of any materials requiring removal.

Project management will be under the direction of Kerr-McGee Chemical Corporation's West Chicago facility management and will include a health physicist. The project management personnel will have the authority to exercise judgment and make decisions to expedite the work.

0620q

Appendix 1

MATERIAL REMOVAL AND RESTORATION AGREEMENT

I am the owner of real property located at

_____

I grant Kerr-McGee Chemical Corporation permission to enter the property named above and remove material and restore the areas from which the material has been removed, generally as described below.

REMOVAL: _____

_____

_____

_____

_____

_____

RESTORATION: Property will be restored to original condition, as per photographs taken of the area, including yard, plantings and/or structures involved in the excavation process. Resident agrees to water all restored areas as necessary. Resident agrees that some plants, shrubs or trees may be replaced with like species younger and smaller in size consistent with nursery husbandry practices.

By: _____

Address: _____

Date: _____

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